Biological Evaluation
For the National Marine Fisheries Service
In regards to the U.S. Virgin Islands
Water Quality Standards

Prepared by:

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Region 2

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PREFACE

This analysis document presents a biological evaluation for the aquatic life water quality standards included in the U.S. Virgin Islands' Water Quality Standards Regulations adopted by the Government of the US Virgin Islands on September 9, 2015. This document includes a presentation and analysis of the available data and the effects determination for eighteen Federally-listed threatened or endangered marine species. This analysis applies the best available scientific and commercial data on what is currently known about the effects of parameters of interest to determine whether the U.S. Virgin Islands' criteria are consistent with levels that do not pose a risk to species listed under the Endangered Species Act.

This Biological Evaluation has been prepared to support the U.S. Environmental Protection Agency's (EPA's) determination of "not likely to adversely affect (NLAA)" the threatened and endangered marine species located in the U.S. Virgin Islands waters covered by the subject water quality standards actions. The adoption of the 2015 water quality standards by the U.S. Virgin Islands Government and EPA approval of this action are considered to be NLAA based on a holistic consideration of the "best available scientific and commercial data."

The EPA views the adoption of numeric water quality criteria as an important step forward for the U.S. Virgin Islands in being able to restore and/or protect the aquatic life within estuaries and coastal environment.

CONTENT

I. Scope of Federal Action

The Endangered Species Act – The Act was signed on December 28, 1973 and replaced the Endangered Species Conservation Act of 1969. It provides for the conservation of species that are endangered or threatened and the conservation of their ecosystems. Species are found to be endangered if they are in danger of extinction throughout all or a significant portion of their range. If species are likely to become endangered in the near future, they are found to be threatened. Presently, approximately 2,215 species are listed as endangered or threatened under the Endangered Species Act (ESA). The National Oceanic and Atmospheric Administration's (NOAA's) National Marine Fisheries Service (NMFS) has jurisdiction over 125 endangered and threatened marine species. Out of 125, a total of 17 species are presently listed (and one is presently being proposed for listing) for the United States Virgin Island's (USVI's) waters. The NMFS works with the U.S. Fish and Wildlife Service (FWS) to manage ESA-listed species. Generally, NMFS manages marine species, while FWS manages land and freshwater species.

Clean Water Act, Section 303(c) - Section 303 of the Clean Water Act (CWA) requires states to modify and improve their water quality standards (WQS) at least once every three years. A Memorandum of Agreement (MoA) between the U.S. Environmental Protection Agency (EPA), FWS and NMFS was signed in 2001 to enhance the coordination under the CWA and the ESA for section 7 consultations. In order to fulfill the goals of this MoA, which was written with the intent to provide efficient mechanisms for improved interagency cooperation, EPA consults with the Services (FWS and NMFS) on newly proposed and/or revised state aquatic life criteria. The agencies agree that it is prudent to examine the aquatic life criteria to determine if they protect ESA-listed species and their critical habitats, and realize the importance of conducting the consultations on new and/or revised state criteria in a timely fashion, so that any state-adopted aquatic life criteria are protective of that state's listed species and their critical habitats. EPA approves state-adopted aquatic life criteria for waters where state water quality standards specify a use protecting aquatic life and the criteria protect their designated uses. Federally-listed aquatic or aquatic-dependent species that have more than limited exposure to these waters are assessed in this biological evaluation (BE).

Description of Federal Action: Adoption of Water Quality Standards - The WQS regulation at 40 CFR Part 131 describes the requirements and procedures for states to develop, adopt, review, revise, and submit WQS as well as requirements and procedures for the EPA to review WQS as authorized by Section 303(c) of the CWA. States have the primary responsibility to develop and adopt WQS to protect their waters. As required by Section 303(c) of the CWA and 40 CFR §131, EPA reviews the WQS that have been adopted by states. State WQS are not considered effective, under the CWA, until approved by the EPA.

The Federal action being evaluated in this consultation is the approval by the EPA of the USVI's WQS as they relate to the protection of water quality, aquatic life and wildlife uses as set forth in the USVI Code of Regulations, Title 12 Chapter 7. The USVI is expected to submit to the EPA for action the revised WQS Regulations by the end of 2015. Once submitted to EPA for review and approval, the Administrator of EPA Region 2 is tasked to approve such standards within

sixty days after the date of submission if the standards are determined to meet the requirements of the CWA. After approval, the standards will become effective, for CWA purposes, for the waters of the USVI.

Overview of WQS - The WQS program is one of the cornerstones of the CWA. Through this program, states set WQS for waters within their jurisdictions. A water quality standard defines the water quality goals of each water body by designating for each one the beneficial use(s). After uses are defined, water quality criteria are adopted to protect each use. WQS also must contain anti-degradation policies designed to protect and maintain the quality and uses of the high quality waters such as Outstanding Natural Resource Waters. Also, under 40 CFR 131.13, states may, at their discretion, include in their WQS general policies affecting the application and implementation of WQS. These types of policies outline how a mixing zone, variance or low-flow policy will be used to implement WQS.

States adopt WQS to protect public health and welfare, enhance the quality of water, and serve the purposes of the CWA. "Serve the purposes of the Act" (as defined in sections 101(a)(2), and 303(c) of the Act) means that WQS are to restore and maintain the chemical, physical and biological integrity of state waters and provide water quality for the protection and propagation of fish, shellfish, and wildlife, and recreation in and on the water. The uses direct the narrative and numeric water quality criteria that will apply for each use.

EPA publishes guidance or other scientific resources to be used by states as recommended guidance in establishing criteria. Section 303(c)(2)(B) of the CWA requires states to adopt numeric criteria for all toxic pollutants for which criteria have been published under Section 304(a). These criteria can be used by the state to establish scientifically defensible water quality criteria. The criteria are also used to determine use attainment of waters through monitoring and assessment. WQS are also used in establishing permit limits, and in the control of non-point source pollution.

II. Background

The USVI's WQS triennial review process was completed in 2004, 2010 and more recently in 2015. The EPA Region 2 initiated informal consultations with Services on all of these three administrative actions. While Region 2 obtained concurrence from FWS on all three actions, consultations with NMFS have never been completed.

Regarding the 2015 revisions to the Water Quality Standards Regulations (WQSR), Region 2 sent letters to Services to initiate ESA informal consultations on July 30, 2014. Region 2 received a concurrence letter from FWS on August 6, 2014. In response to the EPA's above-referenced letter, on November 25, 2014, NMFS responded with the letter informing the Region that the agency has elected to conduct a batched programmatic consultation for the USVI's WQS. This consultation would include the review of the entire set of aquatic life water quality standards adopted by the USVI, to date, and would not be limited only to new and/or revised standards being considered during the 2015 action, as required by law. In summary, based upon the November 25, 2014 letter from NMFS, EPA is expected to demonstrate that it has evaluated all of the aquatic life standards included in the 2015 USVI's WQSR and it has ensured that its

approval action is not likely to jeopardize the continued existence of ESA-listed species, based on preparation and submission of a detailed BE which addresses all applicable standards. This document is intended to meet this expectation.

III. Action Area

The action area includes the entire USVI's Territory since both freshwaters and marine waters are addressed. Because this document is intended to be a basis of the ESA consultation process with NMFS, the aquatic life evaluation included in this document focuses only on threatened and endangered marine species. In the case of sea turtles, in the U.S., NOAA and FWS have joint jurisdiction, with NOAA having the lead in the marine environment and FWS having the lead on the nesting beaches. Therefore, the evaluation of sea turtle species in this BE will focus on addressing threats to marine environment only.

IV. Federally Listed Species and Environmental Threats

The following list of species has been provided to EPA by NOAA to be addressed during this consultation as the complete list of aquatic or aquatic dependent species present in the waters of the USVI and being under the jurisdiction of the NOAA-NMFS Office.

Invertebrates - Corals:

- Elkhorn coral, *Acropora palmata;* listed in 2006
- Staghorn coral, *Acropora cervicornis;* listed in 2006
- Pillar coral, *Dendrogyra cylindrus;* listed in 2014
- Lobed star coral, *Orbicella annularis;* listed in 2014
- Mountainous star coral, *Orbicella faveolata;* listed in 2014
- Boulder star coral, *Orbicella franksi;* listed in 2014
- Rough cactus coral, *Mycetophyllia ferox;* listed in 2014

Mammals - Whales:

- Blue whale, *Balaenoptera musculus;* listed in 1970
- Fin whale, *Balaenoptera physalus;* listed in 1970
- Sei whale, *Balaenoptera borealis;* listed in 1970
- Sperm whale, *Physeter macrocephalus;* listed in 1970
- Humpback whale, *Megaptera novaeangliae;* listed in 1970

Reptiles - Sea Turtles:

- Hawksbill sea turtle, *Eretmochelys imbricata*; listed in 1970

- Green sea turtle, *Chelonia mydas;* listed in 1978
- Loggerhead sea turtle, *Caretta caretta*; listed in 1978
- Leatherback sea turtle, *Dermochelys coriacea;* listed in 1970

Marine Fish:

- Scalloped Hammerhead Shark, *Sphyrna lewini;* listed in 2014
- Nassau Grouper, *Epinephelus striatus;* proposed for listing in 2015

All other threatened or endangered species were determined to be terrestrial and therefore not present in the action area. The FWS have consultation lead on the aquatic species related to the freshwater areas (including nesting beaches for sea turtle species). As it was already mentioned, ESA consultation with FWS on the 2015 USVI's WQSR has been already completed.

V. Environmental threats to the USVI Marine Species

A. General environmental threats to Corals

As described by NOAA, most reef-building stony corals contain in their tissues photosynthetic algae (zooxanthellae), which are responsible for their unique and beautiful colors (NOAA website: http://oceanservice.noaa.gov/education/kits/corals/coral02 zooxanthellae.html). Both, corals and algae, have a symbiotic relationship. Corals provide algae with a protected environment along with compounds they need for photosynthesis. Algae produce oxygen and supply corals with glucose, glycerol, and amino acids, which are the products of photosynthesis, and are used by corals to make proteins, fats, and carbohydrates, and produce calcium carbonate.

When conditions become stressful, corals often expel their algal cells and the colony becomes white in color resulting in "coral bleaching" (Rosenberg and Ben-Haim 2002). When the colored algae are expelled from the coral, the coral not only loses its color but also its source of food. Coral bleaching can have devastating impacts on sensitive coral reef ecosystems, altering community structures caused by increased mortality, lowered reproductive capacity, reduced calcification rates and growth, as well as decreased repair capabilities, which further makes corals more susceptible to disease and other stressors (Hoegh-Guldberg 1999; Rosenberg and Ben-Haim 2002; Fischlin et al. 2007). As reported by Hoegh-Guldberg et al., coral bleaching and resulting mortality becomes progressively worse as stress intensifies and lengthens (Hoegh-Guldberg et al. 2007).

Coral bleaching is a generalized stress response of corals and can be caused by a number of biotic and abiotic factors, including:

- sudden changes (increase or decrease) in water temperatures (Gates et al. 1992; Glynn 1991; Jokiel and Coles 1990; Jones at al. 1998);
- increased solar UV radiation (Gleason and Wellington 1993, Glynn at al. 1992);
- changes in acidity of water (Hoegh-Guldberg 1999);
- increased sedimentation (Jokiel, at al. 2014; Prouty et al. 2014);
- bacterial infections (Jones 1997);
- changes in water salinity (Hoegh-Guldberg and Smith 1989); and
- increased concentrations of pesticides/herbicides (Jones at all. 2003).

While most of the above listed triggers may result in localized bleaching events, mass coral bleaching events occurring at larger scale, most often are triggered by periods of increased sea surface temperatures. Progressing ocean acidification further increases the bleaching effects of thermal stress. In addition, climate change acts synergistically to worsen the effects of other stressors to corals such as disease and predation. Athough most recent research indicates that some of the corals can recover after severe bleaching events, bleaching (in general) is a primary cause of great loss of these sensitive ecosystems (Biello 2015). In general, the impacts to the coral species is expected to worsen as climate change and ocean acidification continue to intensify.

It has been well documented in the scientific literature that for their survival, corals require a fairly narrow range of environmental conditions and are primarily constrained by water temperature, light, salinity and nutrients (Primack 2001; Eakin et al. 2009 and NMFS 2009a). Loss and degradation of coral habitat are the primary reasons for coral species deterioration (Primack 2001). In light of the rapidly changing climate, the ability of corals to recover from severe storms, while facing the combined effects of increasing thermal stress and ocean acidification, can be significantly limited.

Overall, presently most of the corals worldwide are under significant threats from impacts of climate change and anthropogenic sources of pollution resulting in changes in precipitation, rising storm intensities and sea level, increased ocean temperatures and rising ocean acidification (Glynn 1991 and Hoegh-Guldberg 1999). In addition, habitats of many coral species are being degraded due to ship traffic, dredging, coastal development, pollution, and agricultural and land use practices that increase sedimentation and nutrient-loading.

As described by NMFS (77 FR 73220),

"…The Caribbean basin is geographically small and partially enclosed, has high levels of connectivity, and has relatively high human population densities. The wider- Caribbean occupies five million square km of water and has 55,383 km of coastline, including approximately 5,000 islands. Shallow coral reefs occupy approximately 25,000 square km (including ≈2,000 square km within US waters), or about 10 % of the total shallow coral reefs of the world. The amount of non-reefal and mesophotic habitat that could potentially be occupied by corals in the Caribbean is unknown, but is likely greater than the area of shallow coral reefs in the Caribbean.

The Caribbean region has experienced numerous disturbances to coral reef systems throughout recorded human history. Fishing has affected Caribbean reefs since before European contact. Beginning in the early 1980s, a series of basin-scale disturbances has led to altered community states, and a loss of resilience (*i.e.*, inability of corals and coral communities to recover after a disturbance event). Massive, Caribbean wide mortality events from disease conditions of both the keystone grazing urchin *Diadema antillarum* and the dominant branching coral species *Acropora palmata* and *Acropora cervicornis* precipitated widespread and dramatic changes in reef community structure. None of the three important keystone species (*Acropora palmata, Acropora cervicornis,* and *Diadema antillarum*) have shown much recovery over decadal time scales. In addition, continuing coral mortality from periodic acute events such as hurricanes, disease outbreaks, and bleaching events from ocean warming have added to the poor state of Caribbean coral populations and yielded a remnant coral community with increased dominance by weedy brooding species, decreased overall coral cover, and increased macroalgal cover. Additionally, iron enrichment in the Caribbean may predispose the basin to algal growth.

Further, coral growth rates in the Caribbean have been declining over decades. Caribbean-wide meta-analyses suggest that the current combination of disturbances, stressful environmental factors such as elevated ocean temperatures, nutrients and sediment loads, and reduced observed coral reproduction and recruitment have yielded poor resilience, even to natural disturbances such as hurricanes. …"

Overall, in the final listing rule NMFS identified nine threats to the corals that posed either a current or future extinction risk and further classified the threats by importance (79 FR 53851). Primary threats identified include ocean warming, disease, and ocean acidification. Threats of medium to low importance include trophic effects of fishing, sedimentation, nutrient enrichment, sea-level rise, predation, and collection and trade.

B. General environmental threats to Whales

Whales are warm-blooded, air breathing mammals. They can reach lengths of more than 100 feet and weigh up to 200 tons. A thick layer of fat called blubber insulates them from cold ocean waters.

According to NMFS (2015c), presently, whale species are facing numerous threats while they are still recovering from a century of commercial whaling where many species where hunted close to extinction. One of the major stresses to whales is a loss or destruction of their habitat, which is directly linked to increasing human activity within marine environments (De Guise et al. 1995). Every year, an estimated 300,000 whales die because of fishing activities. Fishing with set nets (long nets anchored to the seabed and held up by buoys) is a primary threat. Whales can get tangled in these huge wall-like structures and drown, unable to surface for air.

In addition, fisheries, farm fishing, landfills and shipping channels persistently destroy areas that whales use for feeding, resting and breeding. At the same time, warmer ocean temperatures and melting ice in Polar Regions are progressively changing the ecology of the feeding grounds of

many whale species. Industrial waste and trash (including plastic debris), if ingested, are often a cause for whales' starvation and death. Water pollution including toxic chemicals and heavy metals are often present in polar ecosystems and over time are accumulating up through the food chain.

The recreational uses of marine areas (shore development and increased boat traffic) often drive away whales from feeding, resting and breeding areas. Because whales rely on sound to communicate, hunt and navigate, the underwater noise (produced by ships or during exploration for oil/gas or military activities) may also greatly reduce whales' ability to communicate, hunt and navigate.

C. General environmental threats to Sea Turtles

Sea turtles are cold-blooded, air-breathing reptiles. Their bodies are well adapted to the marine environment. They typically migrate a long distances to nest on beaches. According to NOAA's website (http://www.nmfs.noaa.gov/pr/species/turtles/threats.htm), fishing is one of the major threats for sea turtles. Each year, hundreds of thousands of young and adult and sea turtles are accidentally captured in fisheries. In addition, turtles entangled in fishing gear may drown and often suffer serious injuries to their flippers, jaw or esophagus from hooks, lines or ropes. Fishing dredges (extremely heavy metal frames dragged along the ocean floor) can also crush and entrap turtles, causing death and serious injury. Marine debris is a continuing problem for sea turtles. Sea turtles living in the open ocean environment (e.g., leatherbacks, juvenile loggerheads, and juvenile green turtles) commonly ingest or become entangled in marine debris including plastic bags, balloons, bottles, food wrappers and lost fishing gear.

Marine habitats used by sea turtles can be significantly degraded by environmental contamination from coastal runoff, marina and dock construction, dredging, aquaculture, oil and gas exploration and extraction, increased under water noise and boat traffic. Marine pollution can have serious impacts on sea turtles, their prey and habitats (Israel 2013). When pollution enters the water, it contaminates and kills aquatic plant and animal life that is often food for sea turtles. Oil spills, urban runoff from chemicals, fertilizers and petroleum all contribute to this problem According to Israel, sea turtles are highly contaminated with industrial chemicals and pesticides, which makes them vulnerable to thyroid, liver and neurological damage. Research has shown that some of the contamination (due to PCBs, for example) is passed on to eggs resulting in smaller eggs and weaker hatchlings. New research suggests that pollution in the oceans and in near-shore waters is most likely the most significant trigger for a fibropapillomas, a disease often killing sea turtle species.

According to NOAA, global warming could potentially have an extensive impact on all aspects of a turtle's life cycle, as well as impact the abundance and distribution of food, (http://www.nmfs.noaa.gov/pr/species/turtles/threats.htm). Because sea turtles use both marine and terrestrial habits during their life cycles, the effects of climate change are likely to have a devastating impact on these endangered species. With melting polar ice caps, as the water level begins to rise, the size and amount of nesting beaches decrease. Stronger storms, predicted as a

result of increasing temperatures, will continue to erode coastal habitats. Higher temperatures at the beaches can also adversely affect sea turtle gender ratio. Increasing incubation temperatures could result in more female sea turtles, which reduces reproductive opportunities and decreases their genetic diversity.

Multiple natural and anthropogenic threats to sea turtles are well described by the Sea Turtle Conservancy (http://www.conserveturtles.org/seaturtleinformation). Nesting beaches are significantly degraded by humans and altered by urbanization and development. Many coastal property owners have built sea walls to protect their property from natural erosion. These man-made structures reduce turtles nesting habitat and/or displace turtles to less optimal nesting areas. Beach nourishment is often preferable to armoring, however if it is not done correctly, it can also negatively impact sea turtles. In addition, dredging for the sand to nourish a beach can often cause direct threats to sea turtles and their nearshore marine habitats.

Coastal development is posing a significant threat to sea turtles by introducing the artificial light (http://www.conserveturtles.org/seaturtleinformation). Nesting turtles depend on dark, quite beaches to reproduce successfully. Today's man-made, coastal development results in artificial lighting on the beach, which discourages female sea turtles from nesting. Due to the excessive artificial light, turtles often choose a less-than-optimal nesting spots, which affects the chances of producing a successful nest. Hatchlings follow the light (and reflection) of the moon out to the sea, thus near-shore, inland, lighting can also cause sea turtle hatchlings to become disoriented, forcing them to travel inland where they often die of dehydration and predation.

In the U.S., Services have joint jurisdiction for turtles, with NOAA-NMFS having the lead in the marine environment and FWS having the lead on the nesting beaches. Both federal agencies, along with many state agencies and international partners, have issued regulations to eliminate or reduce threats to sea turtles, while working together to recover their populations.

Over the years, NOAA has implemented numerous measures to reduce sea turtle interactions in fisheries by regulations and permits under the ESA and Magnuson-Stevens Fishery Conservation and Management Act. In 1992, NOAA finalized regulations to require turtle excluder devices (TEDs) in shrimp trawl fisheries to reduce interactions between turtles and trawl gear. Since the early 1990s, the agency has implemented the additional measures including, but not limited to, large circle hooks in longline fisheries, time and area closures for gillnets, and modifications to pound net leaders. Since 1989, the U.S. has prohibited the importation of shrimp harvested in a manner that adversely affects sea turtles.

As described by NOAA, the highly migratory behavior of sea turtles makes them shared resources among many nations (http://www.nmfs.noaa.gov/pr/species/turtles/hawksbill.htm). As a result, the conservation efforts for sea turtle populations in one country may be jeopardized by activities in another. Protecting sea turtles on U.S. nesting beaches and in U.S. waters alone, therefore, may not be fully sufficient to ensure the continued existence of the species.

D. General environmental threats to marine Fish

As described by NMFS in the Technical Memorandum (NMFS 1994), one of the primary concerns related to the marine fish population is the impact of present and future contaminant loading to the marine waters on abundance of the resource species. Pollution including the untreated sewage, garbage, fertilizers, pesticides, industrial chemicals and plastics are significantly impacting fish habitats.

In addition, climate change and resulting ocean acidification disrupts metabolism and other biological functions in marine life. Changes in the ocean's carbon dioxide concentration result in accumulation of carbon dioxide in the tissues and fluids of fish and other marine animals. These impacts can cause a variety of problems for marine animals including difficulty with acid-base regulation, metabolic activity, respiration, and ion exchange, leading to impairment of growth and overall higher mortality rates. Tourism is a significant cause of fish habitat degradation as well, with coastlines being turned into new housing and tourist developments.

VI. Species Description: Status, Life History and Environmental Baseline

A. Species Description: Corals

1. Elkhorn coral, *Acropora palmata*

Taxonomy:

Kingdom: Animalia
Phylum: Cnidaria
Class: Anthozoa
Order: Scleractinia
Family: Acroporidae
Genus: *Acropora*
Species: *palmata*

Historical Information and Conservation-Regulatory Status:

NOAA proposed adding elkhorn coral to the Endangered Species list in May of 2005 (70 FR 24359). Elkhorn coral was listed as threatened under the ESA in the Federal Register (FR) notice dated May 9, 2006 (71 FR 26852). NOAA has designated a critical habitat for elkhorn coral in November 2008 (73 FR 72210). In December 2012, NOAA proposed reclassification of

elkhorn coral to endangered species (77 FR 73220), but determined, in September 2014, that this coral species would remain listed as threatened (79 FR 53852).

As described in more details by NOAA in the FR notice (77 FR73220), the elements that contribute to the status of elkhorn coral are as follows: high vulnerability to ocean warming; ocean acidification and disease; high vulnerability to sedimentation and nutrient over-enrichment; uncommon abundance; decreasing trend in abundance; low relative recruitment rate; narrow overall distribution; restriction to the Caribbean; and inadequacy of regulatory mechanisms.

Species Description:

As described by NOAA, over the last 10,000 years, elkhorn coral has been one of three most important Caribbean corals contributing to reef growth and development and, at the same time, providing essential fish habitat (www.nmfs.noaa.gov/pr/species/invertebrates/ elkhorn coral). Elkhorn coral is a large, branching coral with thick and sturdy antler-like branches. Colonies are fast growing. Branches increase in length by 2-4 inches (in) (5-10 centimeters [cm]) per year, with colonies reaching their maximum size in approximately 10-12 years.

Geographic Boundaries and Spatial Distribution:

As reported by NOAA, elkhorn coral is found on coral reefs in southern Florida, the Bahamas, and throughout the Caribbean (www.nmfs.noaa.gov/pr/species/invertebrates/ elkhorn coral). Its northern limit is Biscayne National Park, Florida, and it extends south to Venezuela. It is not found in Bermuda.

Habitat:

Elkhorn coral was formerly the dominant species in shallow water often found at 3 to 16 feet (ft) (1 to 5 meters [m]) depth, throughout the Caribbean and on the Florida Reef Tract, forming extensive, densely aggregated stands. Coral colonies prefer exposed reef crest and fore-reef environments in depths of less than 20 ft (6 m), although isolated corals may occur to depths of 65 ft (20 m).

On November 26, 2008, NOAA designated critical habitat for elkhorn coral in four specific areas: Florida (1,329 square miles), Puerto Rico (1,383 square miles), and the USVI: St. John/St. Thomas (121 square miles) and St. Croix (126 square miles) (73 FR 72210). As indicated in the background section of this FR notice, these areas support the objective for "substrate of suitable quality and availability" to support successful larval settlement and reattachment of coral fragments.

Life History:

Elkhorn coral reproduce both sexually and asexually. Asexual reproduction results in multiple colonies that are genetically identical while sexual reproduction results in the creation of new genotypes.

The dominant mode of reproduction for elkhorn coral is asexual, with new colonies forming when branches break off of a colony and reattach to the substrate. Asexual reproduction

involves fragmentation, where a small branch of elkhorn coral is carried by waves and currents away from the parent colony and grows into new colonies. Reattachment occurs when either live coral tissue located on the "broken" fragment grows onto suitable substrate or encrusting organisms settle on the dead basal areas of the fragment and cement it to the adjacent substratum (Tunnicliffe 1981). Fragmentation can play a major role in maintaining local populations when sexual recruitment is very limited. The larger size of fragments compared to larvae may result in higher survivorship after recruitment (Jackson 1977). Also, unlike sexual reproduction which is restricted seasonally, fragmentation can take place year-round (Szmant 1986).

Sexual reproduction occurs once, each year (July through September) via broadcast spawning of gametes into the water column. Individual colonies (both, male and female), typically release millions of gametes into the water column and given colony will produce both eggs and sperm. The spawning season for elkhorn corals is relatively short, with gametes released on only a few nights (2 to 6 nights after the full moon) during July, August, or September. Timing of spawning depends on latitude. In the southern Caribbean, for example, spawning can occur as late as October. Some populations may also have two spawning events over the course of two months.

The coral larvae live in the plankton for several days until finding a suitable area to settle, however only very few larvae survive to settle and transform into new colonies. Planktonic larvae experience very high mortality from predation or other factors during their planktonic phase (Goreau et al. 1981). Little is known concerning the settlement patterns of elkhorn coral larvae in the wild. In general, upon proper stimulation, coral larvae settle and metamorphose on appropriate substrates. Like most stony building corals, elkhorn corals require hard, consolidated substrate, including stable, dead coral skeleton, for their larvae to settle upon. Larvae are important as the only phase in the life cycle of elkhorn corals that disperse over long distances, genetically linking populations and providing potential to re-populate depleted areas.

Summary of Major Threats:

According to NOAA, since 1980 elkhorn coral populations have collapsed throughout their range from disease outbreaks, with losses compounded locally by hurricanes as well as increased predation, sedimentation and bleaching due to elevated temperatures (www.nmfs.noaa.gov/pr/species/invertebrates/elkhorncoral).

As stated in the FR (70 FR 24359), seven stressors were identified by NOAA as threats affecting elkhorn corals: natural and anthropogenic abrasion and breakage, sedimentation, persistent elevated temperature, competition, excessive nutrients and sea level rise. Elkhorn coral populations are in danger of extinction due to disease, hurricanes, predation, bleaching, algae overgrowth, sedimentation, as well as water temperature and salinity variation. These threats had been further defined by NOAA in September 10, 2014 and published in the FR notice (79 FR 53852):

> "… Information on threat susceptibilities was interpreted in the proposed rule for *A. palmata'*s vulnerability to threats as follows: high vulnerability to ocean warming, disease, acidification, sedimentation, and nutrient enrichment…

… All information on the susceptibility of *A. palmata* to sedimentation can be summarized as follows. *Acropora palmata* is sensitive to sedimentation due to its poor capability of removing sediment and its high reliance on clear water for nutrition, and sedimentation can cause tissue mortality. We conclude that *A. palmata* is highly susceptible to sedimentation…

… *Acropora palmata* is sensitive to nutrients as evidenced by increased mortality after exposure to raw sewage. We conclude that *A. palmata* is highly susceptible to nutrient enrichment."

2. Staghorn coral, *Acropora cervicornis*

Taxonomy:

Kingdom: Animalia
Phylum: Cnidaria
Class: Anthozoa
Order: Scleractinia
Family: Acroporidae
Genus: *Acropora*
Species: *cervicornis*

Historical Information and Conservation-Regulatory Status:

NOAA proposed adding staghorn coral to the Endangered Species list on May 9, 2005 (70 FR 24359). Staghorn coral was listed as threatened under the ESA in the FR notice dated May 9, 2006 (71 FR 26852). NOAA has designated a critical habitat for staghorn coral in November of 2008 (73 FR 72210). In December of 2012, NOAA proposed reclassification of the staghorn coral to endangered species (77 FR 73220), but determined, in September 2014, that this coral species would remain listed as threatened (79 FR 53852).

According to NOAA, as described in the FR notice (77 FR73220), elements that contribute to the status of staghorn coral are as follows: high vulnerability to ocean warming; ocean acidification and disease; high vulnerability to sedimentation and nutrient over-enrichment; uncommon abundance; decreasing trend in abundance; low relative recruitment rate; narrow overall distribution; restriction to the Caribbean and inadequacy of regulatory mechanisms.

Species Description:

Staghorn coral has been one of three most important Caribbean corals in terms of its contribution to reef growth and fish habitat. Staghorn coral is a branching coral with cylindrical branches which in shape resemble male deer antlers and can grow to over 6.5 ft (2 m). This coral exhibits the fastest growth of all known western Atlantic corals, with branches increasing in length by 4 to 8 in (10 to 20 cm) per year.

Geographic Boundaries and Spatial Distribution:

As reported by NOAA, staghorn coral is found in the Atlantic Ocean, Caribbean Sea, and western Gulf of Mexico [non-U.S. waters] (www.nmfs.noaa.gov/pr/species/invertebrates/staghorn coral). Specifically, staghorn coral is found throughout the Florida Keys, the Bahamas, the Caribbean islands, and Venezuela.

Habitat:

Staghorn coral occur in back reef and fore-reef environments from 0 to 100 ft (0 to 30 m) depth. The upper limit is defined by wave forces, and the lower limit is controlled by suspended sediments and light availability. Fore-reef zones at intermediate depths of 15 to 80 ft (5 to 25 m) were formerly dominated by extensive single species stands of staghorn coral until the mid-1980s.

On November 26, 2008, NOAA designated critical habitat for the staghorn corals and published it in the Federal Register notice (73 FR 72210). Critical habitat was designated in four specific areas: Florida (1,329 square miles), Puerto Rico (1,383 square miles), and the USVI: St. John/St. Thomas (121 square miles) and St. Croix (126 square miles). As indicated in the background section of the November 26, 2008 FR notice, these areas support the objective for "substrate of suitable quality and availability" to support successful larval settlement, recruitment, and reattachment of fragments.

Life History:

As described by NOAA, staghorn coral reproduce both sexually and asexually (www.nmfs.noaa.gov/pr/species/invertebrates/staghorncoral. Similarly to elkhorn coral, the dominant mode of reproduction for staghorn coral is asexual fragmentation, with new colonies forming when branches break off a colony and reattach to the substrate. A broken-off branch may land close to the original colony or be moved a short distance by waves. If the location is favorable, fragments grow into a new colony, expanding and occupying additional area. Reattachment occurs when either live coral tissue located on the fragment grows onto suitable substrate or encrusting organisms settle on the dead basal areas of the fragment and cement it to the adjacent substratum (Tunnicliffe 1981).

Fragmentation results in multiple colonies that are genetically identical while sexual reproduction results in the creation of new genotypes. Fragmentation can play a major role in maintaining local populations when sexual recruitment is very limited. The larger size of fragments compared to larvae may result in higher survivorship after recruitment (Jackson 1977). Also, unlike sexual reproduction which is restricted seasonally, fragmentation can take place year-round (Szmant 1986).

Sexual reproduction occurs via broadcast spawning of gametes into the water column once each year in July, August or September. Individual colonies are both male and female and will release millions of gametes, which are released into the water column. Given colony will produce both eggs and sperm. The spawning season for staghorn corals is relatively short, with gametes released on only a few nights (2 to 6 nights after the full moon) during July through September.

Timing of spawning depends on latitude, occurring in a later month (e.g., October) in the southern Caribbean. Some populations may have two spawning events over the course of two months. Successful recruitment of larvae is the only means by which new genetic individuals enter a population, thereby maintaining or increasing genotypic diversity. Larvae are also important as the only phase in the life cycle of staghorn corals that disperse over long distances, genetically linking populations and providing potential to re-populate depleted areas. The coral larvae live in the plankton for several days until finding a suitable area to settle, but very few larvae survive to settle and metamorphose into new colonies. Some massive coral species reach sexual maturity when their colonies grow to about 4 in (10 cm) in diameter, which occurs when they are about 8 years old.

Summary of Major Threats:

Staghorn coral populations are in danger of extinction due to many factors including: disease, hurricanes, predation, bleaching due temperature variation, algae overgrowth, as well as sedimentation. Some of the most notable factors related to the regulation of water quality are algae and turbidity.

As stated in the FR notice (70 FR 24359), NOAA has identified seven stressors as threats affecting staghorn corals: natural abrasion and breakage, anthropogenic abrasion and breakage, sedimentation, persistent elevated temperature, competition, excessive nutrients and sea level rise. The above mentioned threats are highlighted in more details by NOAA in the FR notice dated September 10, 2014 (79 FR 53852):

> "… Information on threat susceptibilities was interpreted in the proposed rule for A. cervicornis' vulnerabilities to threats as follows: High vulnerability to ocean warming, disease, acidification, sedimentation, and nutrient enrichment…
>
> … Elevated nutrients can cause decreased growth in A. cervicornis. The combined effects of nutrients with other stressors such as elevated carbon dioxide and sedimentation appear to be worse than the effects of nutrients alone, and can cause colony mortality in some combinations. Thus, we conclude that A. cervicornis is highly susceptible to nutrient enrichment.
>
> ….Acropora spp. also appear to be particularly sensitive to shading effects resulting from increased sediments in the water column. Because these corals are almost entirely dependent upon sunlight for nourishment, they are much more susceptible to increases in water turbidity and sedimentation than other species. Increased sediments in the water column, which have been documented to impede larval settlement, can result from, among other things, land run-off, dredging and disposal activities, and major storm events…."

3. Pillar coral, *Dendrogyra cylindrus*

Taxonomy:

Kingdom: Animalia
Phylum: Cnidaria
Class: Anthozoa
Order: Scleractinia
Family: Meandrinidae
Genus: *Dendrogyra*
Species: *cylindrus*

Historical Information and Conservation-Regulatory Status:

On December 7, 2012, NOAA published in the FR a proposed rule to list 82 petitioned coral species as endangered/threatened, including pillar coral (77 FR 73220). On September 10, 2014, NOAA published a final rule to list 20 coral species, including pillar coral as threatened species (79 FR 67356). The listing of the 20 species was effective October 10, 2014.

As described in details by NOAA in the FR notice (77 FR73220), elements that contributed to the listing of the pillar coral are as follows: high vulnerability to disease; moderate vulnerability to ocean warming and acidification; rare general range wide abundance; low relative recruitment rate; narrow overall distribution (based on narrow geographic distribution and moderate depth distribution); restriction to the Caribbean; and inadequacy of regulatory mechanisms.

Species Description:

The pillar coral possesses one of the most distinct morphologies of any coral species, with colonies forming several enormous spires that may reach 2 m in height. These large colonies are typically grey or brown in color and have a hairy appearance when polyps are extended. Despite being offered protection through a number of marine protected areas, this species is threatened by global stressors such as climate change and ocean acidification.

According to NOAA, the *Dendrogyra* genus has only one species, *Dendrogyra cylindrus* (Veron 2000). Colonies are comprised of cylindrical columns up to 2 m high on top of encrusting bases. During the day, gray-brown coral's tentacles typically remain extended. The species is resistant to heavy wave surge but occasionally the base of the colony bioerodes. In such a case, the upper portions of the colonies survive and new pillars are produced and continue to grow upward.

Geographic Boundaries and Spatial Distribution:

Colonies of pillar coral are found in flat, sheltered locations ranging from the southern tip of Florida down to Venezuela and Panama. According to Tunnell (1988), pillar corals have been reported in the waters of south Florida and the U.S. Caribbean. Within Federally protected U.S. waters, the species has been recorded from the following areas: Florida Keys National Marine Sanctuary, Navassa National Wildlife Refuge, Dry Tortugas National Park, USVI's National Park/Monument, Biscayne National Park and Buck Island National Monument. Pillar coral species is widespread, but uncommon throughout its range.

As reported by NOAA, pillar coral occurs throughout the Caribbean and off the southeast coast of Florida. It is uncommon and appears as scattered, isolated colonies, though it is rarely found in aggregations. The species has been described as rare on many Caribbean reefs, and small colonies are unusual (Szmant 1986).

Pillar coral is widely distributed throughout coral reefs of the Caribbean Sea and the subtropical and tropical West Atlantic, ranging from the northern coast of South America (Colombia) to southern Florida (Smith 1971; Veron 2000). Reported distributions on wider Caribbean reefs include: rear zone, reef flat and buttress zone (Goreau 1959); a range of 2 to 20 m, but typically occurring from 3 to 8 m depth (Goreau and Wells 1967); rear zone from 2 to 3 m depth (Pressick 1970); spur-and-groove reefs (14 m) and back reef (1 m) (Cairns 1982); spur-and-groove reefs (Tomascik and Sander 1987); and spur-and-groove reefs (Wheaton and Jaap 1988).

Habitat:

Colonies of pillar coral are found in flat or gentle sloping reefs within more sheltered locations. Most commonly found at depths from 5 to15 m, although they can survive at depths down to 25 m. Because this species propagates by fragmentation, this species thrives in shallower, well-circulated areas.

Life History:

The age of first maturity of most reef building corals is typically 3 to 8 years (Wallace 1999). As such, the average age of mature pillar corals is likely greater than 8 years. The Pillar coral can reproduce both sexually and asexually. Asexual reproduction occurs by fragmentation, when a pillar breaks off the main structure and new pillars grow up from the fallen one.

Sexual reproduction occurs through colonies releasing eggs or sperm into the water where fertilization takes place. Pillar coral individuals are either male or female and release eggs or sperm, respectively, for reproduction. Spawning of sperm and eggs is usually synchronized between colonies to ensure the greatest chance of successful fertilization. Pillar corals are known to spawn approximately 1 to 3 days after each full moon in August (Szmant 1986; Neely et al. 2013). Survivorship of juvenile larvae is thought to be low due to the number of stressors including predation, and competition for light and space.

Total longevity of pillar coral is not known, but it likely to be more than 10 years and may be as long as 100 to 200 years. Annual growth in the northern Keys has been reported to range between 0.56 and 0,9 in (14.3 and 23 milimeters [m]) per year (Hudson et al. 1997).

Summary of Major Threats:

Water quality and climate change (including ocean acidification) are major threats to pillar corals. Bleaching and extensive habitat reduction, due to a combination of threats, both pose significant challenges to this species (IUCN Species Account). In addition to coral bleaching, the frequency and intensity of storms is increasing as the global climate changes, which may directly impact pillar coral ecosystems. Changes in rainfall may also affect sedimentation, salinity, nutrient and pollutant inputs, further impacting these sensitive coral ecosystems (Rogers

1990).

Additionally, the frequency of documented coral disease outbreaks has dramatically risen over the last several decades. An increase in disease occurrences has been connected to deteriorating water quality and increased sea surface temperatures (Peters 1997; Bruno et al. 2007). Pillar corals are particularly sensitive to the white plague and over the years, have suffered partial colony mortality due to this disease, which is caused by bacteria and can progress quickly, spreading 1-2 cm per day.

A direct human impact within pillar corals range increased coastal development as a result of continued human population growth and a demand for tourism. Coastal development can lead to threats such as the run-off of waste water and sewage into the ocean and increased significantly sedimentation, which can decrease light penetration through the water column, reducing photosynthesis and subsequent coral growth.

Another serious threat in many countries in the Caribbean region is over-fishing. This can lead to an algal phase shift, where algae replace coral as the dominant benthic taxa on the reef as a result of a reduction in the number of herbivorous fish that normally feed on the algae.

4. Lobed star coral, *Orbicella* (formerly *Montastraea*) *annularis*

Taxonomy:

Kingdom: Animalia
Phylum: Cnidaria
Class: Anthozoa
Order: Scleractinia
Family: Faviidae
Genus: *Orbicella* (formerly *Montastraea*)
Species: *annularis*

Historical Information and Conservation-Regulatory Status:

On December 7, 2012, NOAA published a Federal Register notice (77 FR 73220) with a proposed rule to list 82 petitioned coral species as endangered/threatened, including lobed star coral. On September 10, 2014, NOAA has published a final rule (79 FR 53852) to list 20 of the coral species, including lobed star coral as threatened species. The listing of the 20 species was effective on October 10, 2014.

As described in more detail by NOAA in the Federal Register notice (77 FR73220), elements that contribute to the status of lobed star coral are as follows: high vulnerability to ocean warming; disease, and ocean acidification; high vulnerability to sedimentation and nutrient over-enrichment; decreasing trend in abundance; low relative recruitment rate; narrow overall distribution (based on narrow geographic distribution and moderate depth distribution; restriction to the Caribbean; and inadequacy of regulatory mechanisms.

Species Description:

Lobed star coral form large, branching colonies of thick columns reaching up to 2 m in length (Weil and Knowlton 1994). Living tissue is generally restricted to the tops of columns. Column sides nearest the live tissue margin have few small polyps that are generally not actively growing. Although columns appear to be very large, only the outer few millimeters represent living tissue, while the rest is a calcium carbonate skeleton. Lobed star coral structures may be hundreds of years old, growing only a few centimeters each year. In some areas, several colonies grow together to form a nearly continuous stretch of lobe star corals that may be tens of meters (or more) long (website: http://oceana.org/marine-life/corals-and-other-invertebrates/lobe-coral).

Geographic Boundaries and Spatial Distribution:

As reported by NOAA, lobed star coral is found throughout the Caribbean Sea, including in the Bahamas, Bermuda and Flower Garden Banks (77 FR 73220). The range is restricted to the West Atlantic and there is no range fragmentation. Within Federally protected waters, this species has been recorded from the following areas: Flower Garden Banks National Marine Sanctuary, Dry Tortugas National Park, the USVI's National Park/Monument, Florida Keys National Marine Sanctuary, Navassa Island National Wildlife Refuge, Biscayne National Park and Buck Island Reef National Monument.

Habitat:

Like most shallow-water corals, lobed star corals have symbiotic algae living within their cells, providing the corals with excess energy that they make via photosynthesis (http://oceana.org/marine-life/corals-and-other-invertebrates/lobe-coral). Nearly all species of shallow-water corals and several other groups of reef invertebrates have symbiotic relationships with these algae, so it is important that they live in clear, shallow water. Lobed star corals also filter feed and eat small zooplankton and other prey from the water column. This food provides them with additional energy and provides their symbiotic algae with the necessary nutrients to continue to generate food.

No critical habitat rules have been published for the lobed star coral (http://ecos.fws.gov/speciesProfile/profile/speciesProfile/spcode=P003#crithab).

Life History:

Each structure of the lobed star coral is actually a colony of several genetically identical animals living together. Unlike many species of corals, lobed star corals are either male or female, not both. They reproduce via broadcast spawning, where several individuals release their eggs or sperm into the water column at the same time. This method increases the likelihood that eggs become fertilized and reduces the danger from egg predators near the reef surface. Within a few days after the eggs hatch, larvae settle onto the reef surface and begin to form new colonies.

Summary of Major Threats:

According to the International Union for the Conservation of Nature (IUCN), lobed star coral is believed to decline by 50% or more in 30 years, due to anthropogenic factors (IUCN Species Account). Specifically, this species has suffered a severe decline in the overall cover and abundance in several parts of the Caribbean, including cover losses of 40-60% off the south and southeast coasts of Puerto Rico (E. Weil, personal communication, in IUCN Species Account) and cover losses of 72% off of St. John (Edmunds and Elahi 2007; IUCN Species Account).

Reported threats to lobed star coral include climate-related ocean acidification and bleaching, infectious diseases, predation by stoplight parrotfish (*Sparisoma viride*), hurricane damage, loss of habitat from algal overgrowth and sedimentation, localized bioerosion by sponges and other organisms, and other diseases (IUCN Species Account). According to the IUCN, current rates of mortality are exceeding growth and recruitment and the current threats are continuing to increase.

According to NOAA, elements that contribute to lobed star coral status are: high vulnerability to ocean warming; disease, and ocean acidification; high vulnerability to sedimentation, and nutrient over-enrichment; decreasing trend in abundance; low relative recruitment rate; narrow overall distribution (based on narrow geographic distribution and moderate depth distribution); restriction to the Caribbean and inadequacy of regulatory mechanisms (77 FR 73220).

5. Mountainous star coral, *Orbicella (formerly Montastraea) faveolata*

Taxonomy:

Kingdom: Animalia
Phylum: Cnidaria
Class: Anthozoa
Order: Scleractinia
Family: Faviidae
Genus: *Orbicella (formerly Montastraea)*
Species: *faveolata*

Historical Information and Conservation-Regulatory Status:

On December 7, 2012, NOAA published a FR notice including a proposed rule to list 82 petitioned coral species as endangered/threatened, including mountainous star coral (77 FR 73220). On September 10, 2014, NOAA published a final rule (79 FR 53852) to list 20 of coral species, mountainous star coral including, as threatened species. The listing of the 20 species was effective October 10, 2014.

As described by NOAA, elements that contribute to the status of mountainous star coral are as follows: high vulnerability to ocean warming disease, and ocean acidification; high vulnerability to sedimentation and nutrient over-enrichment; decreasing trend in abundance; low relative recruitment rate; moderate overall distribution (based on narrow geographic distribution and wide depth distribution, restriction to the Caribbean and inadequacy of regulatory mechanisms (77 FR73220).

Species Description:

Mountainous star coral has been called the "dominant reef-building coral of the Atlantic" (Smith et al. 2006). Buds of the mountainous star coral extend to form head or sheet colonies with corallites that are uniformly distributed and closely packed (Weil and Knowlton 1994). Septa are highly exert, with septocostae arranged in a variably conspicuous fan system, and the skeleton is generally far less dense than those of its sibling species. Active growth is typically found at the edges of colonies, forming a smooth outline with many small polyps. In the FR notice (79 FR 53852), NOAA reported growth rates of mountainous star coral ranging between 0.3 and 1.6 cm per year (Cruz-Piñón et al. 2003).

Geographic Boundaries and Spatial Distribution:

Mountainous star coral occurs in the Caribbean, the Gulf of Mexico, Florida and the Bahamas. Within Federally protected waters, the species has been recorded from the following areas: Flower Garden Banks National Marine Sanctuary, Florida Keys National Marine Sanctuary, Biscayne National Park, Dry Tortugas National Park, Virgin Islands National Park/Monument, Navassa Island National Wildlife Refuge and Buck Island National Monument.

Habitat:

According to the IUCN (Red List of Threatened Species, 2015, available at http://www.iucnredlist.org/details/133373/0), mountainous star coral is found from 1- to 30 m in back reef and fore-reef habitats, and is often the most abundant coral between 10- and 20 m in fore-reef environments. The range of depth for mountainous star coral is similar but broader than lobe star coral, with significant overlap. It is more aggressive than lobe star coral, but less aggressive than boulder star coral, described below.

Life History:

Like most other coral species, mountainous star coral use both sexual and asexual propagation. Sexual reproduction is primarily observed through gametogenesis (i.e., development of eggs and sperm within the polyps near the base). Asexual reproduction involves fragmentation, where colony pieces or fragments are dislodged from larger colonies to establish new colonies.

Depending on the mode of fertilization, coral larvae (called planulae) undergo development either within or outside of the mother colony. In either mode of larval development, as in case of other coral species, larvae experience considerable mortality, reaching as high as up to 90 %, primarily from predation or other factors prior to settlement and metamorphosis. Coral larvae are relatively poor swimmers; therefore, their dispersal distances largely depend on the duration

of the pelagic phase and the speed and direction of water currents transporting the larvae (77 FR 73223).

Summary of Major Threats:

Like lobed star coral, mountainous star coral is listed by the IUCN as endangered because it is believed to have declined by 50 % or more over 30 years (IUCN Species Account). Mountainous star coral has experienced comparable cover losses in Jamaica, Puerto Rico, St. John, and Carysfort Reef. Mountainous star coral faces the same threats as lobed star coral (described in sections above). These threats are increasing and spreading into new areas (IUCN Species Account). Current rates of mortality are exceeding growth and recruitment, and the chances of recovery are limited due to the species' extreme longevity, low recruitment rates, and long generation times. A study of *O. faveolata* colonies in the Florida Keys during and after the 2005 mass bleaching event found that corals with greater bleaching intensities later developed white plague infections (Brandt and McManus 2009), suggesting that this species is susceptible to loss of disease resistance during intense bleaching events.

According to NOAA (79 FR 53852), the major threats to this species include: high vulnerability to ocean warming, disease, acidification, sedimentation, and nutrient enrichment; moderate vulnerability to the trophic effects of fishing; and low vulnerability to sea level rise, predation, and collection and trade.

6. Boulder star coral, *Orbicella (formerly Montastraea) franksi*

Taxonomy:

Kingdom: Animalia
Phylum: Cnidaria
Class: Anthozoa
Order: Scleractinia
Family: Faviidae
Genus: *Orbicella (formerly Montastraea)*
Species: *franksi*

Historical Information and Conservation-Regulatory Status:

On December 7, 2012, NOAA published a proposed rule to list 82 petitioned coral species as endangered/threatened, including boulder star coral (77 FR 73220). On September 10, 2014, NOAA published a final rule to list 20 proposed coral species, including boulder star coral, as threatened species (79 FR 53852). The listing of the 20 species was effective on October 10, 2014.

As described in detail by NOAA in the FR notice (77 FR73220), elements that contribute to the status of boulder star coral are as follows: high vulnerability to ocean warming disease, and ocean acidification; high vulnerability to sedimentation and nutrient over-enrichment; decreasing trend in abundance; low relative recruitment rate; moderate overall distribution (based on narrow geographic distribution and wide depth distribution, restriction to the Caribbean; and inadequacy of regulatory mechanisms.

Species Description:

Boulder star coral builds massive, encrusting plate or subcolumnar colonies via extratentacular budding (Weil and Knowlton 1994). The characteristically bumpy appearance of this species is caused by relatively large, unevenly exert, and irregularly distributed corallites. Boulder star coral is distinguished from its sibling Orbicell species by this irregular or bumpy appearance; a relatively dense, heavy, and hard skeleton (corallum) and a greater degree of interspecies aggression.

Geographic Boundaries and Spatial Distribution:

Boulder star coral is known to occur in the Caribbean, Gulf of Mexico, Florida, the Bahamas and Bermuda (IUCN Species Accounts). The star corals in the Orbicella species complex historically dominated coral reefs throughout the Caribbean both by abundance and cover. Over the last twenty years, major declines between 50 and 95 % have been reported in many locations; a few locations report stable or increasing coverage. Since the 1980's decline of Acropora spp., total coral cover decline in the Caribbean has been associated with the decline of the star corals.

Star corals (Orbicella spp.) have slow growth rates, late reproductive maturity, and low recruitment rates (Bruckner, 2012). Colonies can grow very large and live for centuries. Partial mortality of large colonies is common on modern reefs. These large colonies of star corals have been able to maintain populations over time, but recent population declines and partial colony mortality is resulting in smaller colonies with less reproductive output and even lower replenishment potential. The historical presence of few small colonies coupled with observation of few recruits in the presence of large gamete production from the large colonies suggests recruitment events are rare, and were less important for the survival of the Orbicella species complex in the past than today (Bruckner, 2012).

Habitat:

According to the IUCN (Red List of Threatened Species, 2015 (available at: http://www.iucnredlist.org/details/133373/0*)*, this is a common species. Boulder star coral is found from 5 to 50 m, and is often the most abundant coral from 15 to 30 m in fore-reef environments (Weil and Knowlton 1994; Szmant *et al*. 1997).

Life History:

Like most other coral species, boulder star coral use both sexual and asexual propagation. Sexual reproduction is primarily observed through gametogenesis (i.e., development of eggs and sperm within the polyps near the base). Asexual reproduction involves fragmentation, where colony pieces or fragments are dislodged from larger colonies to establish new colonies.

The age of first maturity of most reef building corals is typically three to eight years (Wallace 1999) and the average age of mature individuals is greater than eight years (IUCN Red List of Threatened Species, 2015 (available at: http://www.iucnredlist.org/details/133373/0*)*. Furthermore, based on average sizes and growth rates reported for boulder star coral, the average generation length for this species is estimated to be 10 years.

Summary of Major Threats:

The threats faced by boulder star coral are the same as those faced by lobed star coral, detailed above (IUCN Species Account). Boulder star coral species has historically shown greater resistance to disease than its siblings, but the past 10 years have seen significant declines, with accelerating losses of cover in U.S. waters since 2002. Boulder star coral is listed as vulnerable by the IUCN due to these recent trends and the associated increased threat susceptibility (IUCN Species Account). Vulnerability to disease and habitat degradation increases the likelihood of the species being lost within one generation, and the species is projected to lose 38% of its population over next 30 years.

7. Rough cactus coral, *Mycetophyllia ferox*

Taxonomy:

Kingdom: Animalia
Phylum: Cnidaria
Class: Anthozoa
Order: Scleractinia
Family: Faviidae
Genus: *Mycetophyllia*
Species: *ferox*

Historical Information and Conservation-Regulatory Status:

On December 7, 2012, NOAA published a proposed rule (77 FR 73220) to list 82 petitioned coral species as endangered/threatened, including rough cactus coral. On September 10, 2014, NOAA published a final rule (79 FR 53852) to list 20 of the proposed coral species, including rough cactus coral, as threatened species. The listing of the 20 species was effective on October 10, 2014.

As well described by NOAA (77 FR73220), elements that contribute to the status of rough cactus coral are as follows: high vulnerability to disease; moderate vulnerability to ocean warming and acidification; high vulnerability to nutrient over-enrichment; rare general rangewide abundance; decreasing trend in abundance; low relative recruitment rate; moderate overall distribution (based on narrow geographic distribution and wide depth distribution, restriction to the Caribbean and inadequacy of regulatory mechanisms.

Species Description:

Colonies of the genus *Mycetophyllia* consist of flat plates with radiating valleys (Veron 2000). Rough cactus coral is a widely recognized valid species with colonies comprised of thin, weakly attached plates with interconnecting, slightly sinuous, narrow valleys. Tentacles are generally absent in species of this genus except at the margins of colonies. Corallite centers tend to form single rows, and columellae, when present, are rudimentary. Valleys and walls are contrasting shades of grays and browns. While rough cactus coral is most abundant in fore-reef environments at depths of 10 to 20 m, it is also found in a broader range of habitats including deeper back reefs and lagoons (IUCN Species Account).

Geographic Boundaries and Spatial Distribution:

The range of rough cactus coral is restricted to the West Atlantic. There it has been reported to occur throughout most of the Caribbean, including the Bahamas, but it is not present in the Flower Garden Banks or around the waters of Bermuda. Within Federally protected waters, this species has been recorded from the following areas: Dry Tortugas National Park, Virgin Island National Park/Monument, Florida Keys National Marine Sanctuary, Navassa Island National Wildlife Refuge, Biscayne National Park and Buck Island Reef National Monument. Rough cactus coral is the most dominant species of the *Mycetophyllia* genus in shallow and intermediate depths throughout its range, which includes the Caribbean, southern Gulf of Mexico, Florida, and the Bahamas (IUCN Species Account).

Habitat:

According to the IUCN Red List of Threatened Species, 2015 (http://www.iucnredlist.org/details/133373/0*)*, rough cactus coral species is most common in fore-reef environments from 5-30 meters (but is more abundant from 10-20 m), but also occurs at low abundance in certain deeper back reef habitats and deep lagoons.

Life History:

Like most other coral species, rough cactus coral use both sexual and asexual propagation. Sexual reproduction is primarily observed through gametogenesis (i.e., development of eggs and sperm within the polyps near the base). Asexual reproduction involves fragmentation, where colony pieces or fragments are dislodged from larger colonies to establish new colonies. The species has the potential to exhibit recovery, because of its reproductive strategy (e.g., brooding with moderate recruitment success).

According to the IUCN Red List of Threatened Species, 2015 (available at: http://www.iucnredlist.org/details/133373/0), the age of first maturity of most reef building corals is typically three to eight years (Wallace 1999) and the average age of mature individuals is greater than eight years. Furthermore, based on average sizes and growth rates, the average generation length is 10 years, unless otherwise stated. Total longevity is not known, but likely to be more than ten years.

Summary of Major Threats:

Rough cactus coral has suffered significant localized declines throughout its range due to disease and bleaching (IUCN Species Account). The first outbreaks of white plague were in Florida in 1975 and the 1980s, from which the species made a partial unexpected recovery with documented new recruits (Dustan and Halas 1987; IUCN Species Account).

Subsequent outbreaks throughout the Caribbean since the 1990s have been increasingly virulent and have caused significant mortality throughout rough cactus coral colonies (IUCN Species Account). A 2005 bleaching event caused high rates of mortality off Puerto Rico and its associated islands as well as off Grenada. Rough cactus coral is also susceptible to black band disease and sedimentation, especially when it is already compromised by white plague or bleaching. The IUCN lists rough cactus coral as vulnerable due to its recent increased threat susceptibility and the estimated loss of 38% of the population within next 30 years.

B. Species Description: Whales

1. Blue whale, *Balaenoptera musculus*

Taxonomy:

Kingdom: Animalia
Phylum: Chordata
Class: Mammalia
Order: Cetacea
Family: Balaenopteridae
Genus: *Balaenoptera*
Species: *musculus*

Historical Information and Conservation-Regulatory Status:

According to NOAA (http://www.fisheries.noaa.gov/pr/species/mammals/whales/blue-whale.html), the blue whale is listed as endangered throughout its range under the ESA, and, thus, is listed as "depleted" throughout its range under the Marine Mammal Protection Act. In June of 1970, the blue whale, along with all of the other baleen whales and the sperm whale, was placed on the list of endangered species (35 FR 8491).

Internationally, blue whales received complete legal protection from commercial whaling in 1966 under the International Convention for the Regulation of Whaling. In 1998, NOAA published a Blue Whale Recovery Plan. The Plan details the comprehensive and long-term conservation efforts for blue whales. In April 2012, NOAA announced that it intends to update the recovery plan for the blue whale and requested comments and information from the public.

Conservation actions for the blue whale are ongoing and include: monitoring the status of the Eastern North Pacific Stock of blue whales via shipboard surveys; implementing a number of ship strike reduction measures in southern and central California; placing observers onboard vessels to monitor the take of protected species, including other marine mammals; and implementing marine mammal take reduction measures identified in the Pacific Offshore Cetacean Take Reduction Plan to reduce the bycatch of blue whales and other marine mammals.

Species Description:

Blue whales are the largest animals ever known to live in Earth. They are marine mammals, representing a suborder of baleen whales (http://www.fisheries.noaa.gov/pr/species/mammals/whales/blue-whale.html).

Unlike toothed whales, baleen whales are born with baleen plates instead of teeth (Mizroch et al. 1984). Due to the absence of teeth, baleen whales hunt for their food using a method known as filter feeding. As well described by NOAA, these whales capture their food by swimming towards their prey, with their mouth open, and use their baleen bristles to filter large amounts of fish, krill, shrimp, octopus, various crustaceans and other sea sediments from the water. They then expel the water out of their mouth, while leaving their prey stuck in the baleen bristles, before it is swallowed.

In the North Atlantic and North Pacific, blue whales can grow up to about 90 ft (27 m), but in the Antarctic, they can reach up to about 110 ft (33 m) and can weigh more than 330,000 pounds (lbs) (150,000 kilograms [kg]). Like other baleen whales, female blue whales are somewhat larger than males.

Blue whales have a long-body and comparatively slender shape, which allows they to swim up to 25 miles per hour (mph). They have a gray color pattern that appears light blue when seen through the water (hence, the "blue" whale).

In terms of social structure baleen whales are known to be solitary in nature, often traveling alone. During migration, feeding or breeding periods they tend to form larger groups. Communication among baleen whales involves loud low-pitched moans and whines. The blue whale is one of the loudest animals in existence, which can be heard several miles away and far below the ocean's surface.

Geographic Boundaries and Spatial Distribution:

Blue whales are baleen whales and are found worldwide. According to NOAA, blue whales are found in all oceans and are separated into populations by ocean basin in the North Atlantic, North Pacific, and Southern Hemisphere (http://www.fisheries.noaa.gov/pr/species/mammals/whales/blue-whale.html). In the North Atlantic Ocean, the range of blue whale extends from the subtropics to the Greenland Sea. Blue whales are most frequently sighted in the waters off eastern Canada, with the majority of recent records from the Gulf of St. Lawrence, where they are present throughout most of the year. They are most common during the summer and fall feeding seasons and typically leave by early winter. Although they are rare in the shelf waters of the eastern U.S., blue whales are

occasionally seen off Cape Cod, MA. It is believed this region may represent the current southern limit of the blue whales' feeding range. In addition, some evidence suggests that blue whales occur infrequently in the Gulf of Mexico and the Caribbean.

They migrate seasonally between summer and winter, but some evidence suggests that individuals remain in certain areas year-round. Information about distribution and movement of this species vary with location, and its migratory routes are not well known. In general, distribution is driven largely by food requirements and they occur in waters where krill is concentrated. These whales migrate towards colder polar waters during feeding season when large abundances of krill inhibit the cool waters and will travel to the warmer tropical waters during mating season where they can reproduce and give birth in steady waters. During their migration trips the blue whale can travel thousands of miles from one location to the next. While they migrate most whales will not eat any food and live primarily off of blubber/body fat and stored calories.

Habitat:

Blue whales are found worldwide, from sub-polar to sub-tropical latitudes (http://www.fisheries.noaa.gov/pr/species/mammals/whales/blue-whale.html). Poleward movements in spring allow the whales to take advantage of high zooplankton production in summer. Although blue whales are found in coastal waters, they are thought to occur generally more offshore than other whales.

Life History:

According to NOAA, not much is known about the life history and reproduction of the blue whale (http://www.fisheries.noaa.gov/pr/species/mammals/whales/blue-whale.html). The best available science suggests that the average gestation period for a female blue whale usually lasts 10 to 12 months once the female becomes impregnated. Every 2 to 3 years, the female will give birth to a single offspring, which can measure 20 to 25 ft in length, when born. Baby whale is nursed by the mother for the first 6 to 9 months. Once the young whale matures around the ages of 5 to 10 years, it can begin mating and reproducing on its own. As with other baleen whale species when the blue whale reaches adulthood the female whales typically grow to be larger than their male counterparts. In terms of lifespan it is estimated that a blue whale may live for up to 90 years.

Summary of Major Threats:

As described by NOAA (http://www.fisheries.noaa.gov/pr/species/mammals/whales/blue-whale.html), the primary threats currently facing blue whales are:

- vessel strikes causing mortality and serious injury;
- fisheries interactions through the incidental take are of great concern due to interaction with fishing gear and underestimated entanglement rates. Blue whales may break through or carry away fishing gear, perhaps suffering unrecorded subsequent mortalities or serious injuries. It is also likely that stranding data underestimate the number of

whales killed by fishing gear, because most whales do not drift far enough to strand on beaches or to be detected floating nearshore. Direct observation of mortality is rare;

- habitat degradation/pollution, where the chemical pollution has occurred in some areas of the North Atlantic, but the impacts of this degradation on whales in general are understudied. There is a little evidence available to describe or quantify the specific impacts of this threat on blue whales (O'Shea and Brownell 1994); and
- vessel disturbance. While anthropogenic noise may threaten other cetaceans, little is known about whether, or how, vessel noise affects blue whales. Vessel disturbance (like whale-watching boats) may affect blue whales, but there is no direct evidence to demonstrate that persistent close approaches by tour boats has a negative effect on this species.

2. Fin whale, *Balaenoptera physalus*

Taxonomy:

Kingdom: Animalia
Phylum: Chordata
Class: Mammalia
Order: Cetacea
Family: Balaenopteridae
Genus: *Balaenoptera*
Species: *physalus*

Historical Information and Conservation-Regulatory Status:

In June of 1970, the fin whale, along with all of the other baleen whales and the sperm whale, was placed on the list of endangered species (35 FR 8491).

According to NOAA (http://www.nmfs.noaa.gov/pr/species/mammals/cetaceans/finwhale.htm), the "Atlantic Large Whale Take Reduction Team" was established to develop a plan to reduce the incidental serious injury and mortality of fin whales, right whales, humpback whales, and minke whales in the South Atlantic shark gillnet fishery, the Gulf of Maine and Mid-Atlantic lobster trap/pot fishery, the Mid-Atlantic gillnet fishery, and the Gulf of Maine sink gillnet fishery. On August 5, 1998, NOAA published the draft Recovery Plan for Fin and Sei Whales (63 FR 41802). Within the U.S., the fin whale is listed as endangered throughout its range under the Endangered Species Act of 1973 and is listed as "depleted" throughout its range under the Marine Mammal Protection Act of 1972.

Species Description:

Fin whales are the second-largest, baleen species of whale, with a maximum length of about 75 ft (22 m) in the Northern Hemisphere, and 85 ft (26 m) in the Southern Hemisphere

(http://www.nmfs.noaa.gov/pr/species/mammals/cetaceans/finwhale.htm). Adults can weigh between 80,000 and 160,000 lbs (40 and 80 tons), with lifespan of 80 to 90 years.

Fin whales have a sleek, streamlined body with a V-shaped head. Two thirds of the way down their back, fin whales have characteristic, tall hook shaped fin. The species has a distinctive coloration pattern: the back and sides of the body are black or dark brownish-gray, and the ventral surface is white. The unique, asymmetrical head color is dark on the left side of the lower jaw, and white on the right side. Many individuals have several light-gray, V-shaped "chevrons" behind their head, and the underside of the tail flukes is white with a gray border.

Fin whales can be found in social groups of 2 to7 whales and in the North Atlantic are often seen feeding in large groups that include humpback whales, minke whales, and Atlantic white-sided dolphins (Jefferson et al. 2008). The fin whale can dive for up to 20 minutes at a time, and to depths reaching 1800 ft The fin whale is the fastest swimming of all the large whales and is sometimes referred to as the "greyhound of the seas." Fin whales can swim at up to 30 mph (48 kilometers/hour [km/h]) in short bursts when alarmed and at up to 18 mph (30 km/h) when migrating and cruising.

Similar to other baleen whales, during the summer, fin whales feed on krill, small schooling fish (e.g., herring, capelin, and sand lance), and squid by lunging into schools of prey with their mouth open, using their 50 to 100 accordion-like throat pleats to gulp large amounts of food and water. They then filter the food particles from the water using the 260 to 480 plates on each side of the mouth. Fin whales fast in the winter while they migrate to warmer waters.

Fin whales, similar to other baleen species, are found most often alone, but groups of 3- to 7 individuals are common, and association of larger numbers may occur in some areas at times. Because their powerful sounds can carry vast distances, fin whales may stay in touch with each other over long distances.

Geographic Boundaries and Spatial Distribution:

As reported by NOAA, there are two named subspecies of fin whale: *B. physalus physalus* in the North Atlantic and *B. physalus quoyi* in the Southern Ocean (http://www.nmfs.noaa.gov/pr/species/mammals/cetaceans/finwhale.htm). There is also a population of fin whales in the North Pacific, which most experts consider a separate, unnamed subspecies. These populations rarely mix, if at all, and there are geographical stocks within these ocean basins. Fin whales are migratory, moving seasonally into and out of high-latitude feeding areas, but the overall migration pattern is complex, and specific routes have not been documented. There may be resident groups of fin whales in some areas, such as the Gulf of California, the East China Sea, and the Mediterranean Sea.

Habitat:

Fin whales are found in deep, offshore waters of all major oceans, primarily in temperate to polar latitudes, and less commonly in the tropics (http://www.nmfs.noaa.gov/pr/species/mammals/cetaceans/finwhale.htm). They occur year-

round in a wide range of latitudes and longitudes, but the density of individuals in any one area changes seasonally. No critical habitat rules have been published for the fin whale.

Life History:

Little is known about the social and mating systems of fin whales. Similar to other baleen whales, long-term bonds between individuals are rare. Males become sexually mature at 6- to 10 years of age; females at 7 to 12 years of age. Physical maturity is attained at approximately 25 years, for both sexes.

Females are thought to give birth only at 3-year intervals. Mating and calving occur from November to March in temperate waters. The gestation period is approximately 11 to 12 months and newborn single calfin is about 18 ft (6.4 m) in length and weigh 4,000 to 6,000 lbs (2 tons). The period of lactation lasts from 6 to 7 months and after weaning the young whales are approximately 12.2 m long. As with other migratory baleen whales, northern and southern hemisphere populations do not interbreed due to asynchronous seasons.

Summary of Major Threats:

As reported by NOAA, there is a several threats which contribute to the listing of the fin whale species as threatened:

- historical commercial whaling;
- frequent collisions with vessels (of all species of large whales, fin whales are most often reported as hit by vessels);
- entanglement in fishing gear;
- reduced prey abundance due to overfishing;
- habitat degradation; and
- disturbances from low-frequency noise.

Schooling fish constitute a large proportion of the fin whale's diet in many areas of the North Atlantic, so trends in fish populations, whether driven by fishery operations, human-caused environmental deterioration, or natural processes, may also strongly affect the size and distribution of fin whale populations.

3. Sei whale, *Balaenoptera borealis*

Taxonomy:

Kingdom: Animalia
Phylum: Chordata
Class: Mammalia
Order**:** Cetacea
Family**:** Balaenopteridae
Genus**:** *Balaenoptera*
Species**:** *borealis*

Historical Information and Conservation-Regulatory Status:

This species became important to the whaling industry as populations of blue and fin whales declined. Its earliest exploitation began in the 17th century, off northern Japan. Hunting began in the North Atlantic in the 1800's.

Sei whale species is protected under the Marine Mammal Protection Act of 1972 and the ESA of 1973, as amended. The sei whale did not have meaningful international protection until 1970, when the International Whaling Commission (IWC) first set catch quotas for the North Pacific for individual species (Allen 1980). Before quotas, there were no legal limits. Complete protection from commercial whaling in the North Pacific came in 1976. Quotas on sei whales in the North Atlantic began in 1977 (Reeves et.al. 1998). Southern Hemisphere stocks were protected in 1979. Facing mounting evidence that several whale species were threatened with extinction, the IWC established a complete moratorium on commercial whaling beginning in 1986.

In June of 1970, the sei whale, along with all of the other baleen whales and the sperm whale, was placed on the list of endangered species (35 FR 8491). In 1972, stocks in the North Pacific were estimated to be only 21% of original numbers. According to the Cetacean and Turtle Assessment Program (1982), there may be as few as 2,200 to 2,300 individuals in U.S. Atlantic waters currently. On August 5, 1998, NOAA published the draft Recovery Plan for Fin and Sei Whales (63 FR 41802). NOAA published a final recovery plan for the sei whale in December 2011. The species remained listed on the IUCN's Red List of Threatened Species in 2000, categorized as endangered (Reilly et.al. 2008).

Species Description:

This species of whale is often found with Norway Pollock (*Theragra finnmarchica),* which is a rare fish of the codfish family. In Norway, the name "sei" comes from the Norwegian word for pollock, "seje", where the name for that species of whales originates.

As described by NOAA, sei whales are members of the baleen whale family and are considered one of the "great whales" (http://www.nmfs.noaa.gov/pr/species/mammals/cetaceans/seiwhale.htm). Two subspecies of sei whales are recognized, *B. borealis* in the Northern Hemisphere and *B. schlegellii* in the Southern Hemisphere. The oldest sei whale recorded was 65 years old (Gambell 1985). These large animals can reach lengths of about 40 to 60 ft (12 to 18 m) and weigh 100,000 lbs (45,000 kg), making it the third largest baleen whale, behind the blue and fin whales (Perry et al. 1999). Females may be slightly longer than males. Sei whales have a long, sleek body that is dark bluish-gray to black in color and pale underneath. It can be identified by its inverted "V" shaped water spout which reaches 6 to 8 ft into the air. This whale may be the fastest of the large whales, able to cruise at 16 mph, with a maximum speed of 40 mph recorded.

Sei whales are usually observed singly or in small groups of 2 to 5 animals, but are occasionally found in larger (30 to 50) loose aggregations. Sei whales are capable of diving 5 to 20 minutes to opportunistically feed on plankton (e.g., copepods), krill, small schooling fish, and

cephalopods (e.g., squid) by both gulping and skimming. They prefer to feed at dawn and may exhibit unpredictable behavior while foraging and feeding on prey.

Geographic Boundaries and Spatial Distribution:

The sei whales inhabit most oceans and adjoining seas, and prefer deep offshore waters (Gambell 1985). It avoids polar and tropical waters and semi-enclosed bodies of water. The sei whales migrate annually from cool and subpolar waters in summer to winter in temperate and subtropical waters (Reeves et al. 1998). They prefer temperate waters in the mid-latitudes, and can be found in the Atlantic, Indian, and Pacific Oceans. During the summer, they are commonly found in the Gulf of Maine, and on Georges Bank and Stellwagen Bank in the western North Atlantic. The entire distribution and movement patterns of this species is not well known. This species may unpredictably and randomly occur in a specific area, sometimes in large numbers. These events may occur suddenly and then not occur again for long periods of time.

In the North Atlantic, its range extends from southern Europe or northwestern Africa to Norway, and from the southern United States to Greenland (Gambell 1985). The southernmost confirmed records are strandings along the northern Gulf of Mexico and in the Greater Antilles. Throughout its range, the whale tends to avoid semi-enclosed bodies of water, such as the Gulf of Mexico, the Gulf of Saint Lawrence, Hudson Bay, the North Sea, and the Mediterranean Sea.

Habitat:

Sei whales prefer subtropical to subpolar waters on the continental shelf edge and slope worldwide (Gambell 1985). They are usually observed in deeper waters of oceanic areas far from the coastline. Sei whales are found in the North Atlantic Ocean ranging from Iceland south to the northeastern Venezuelan coast, and northwest to the Gulf of Mexico. There are also records from Cuba and the Virgin Islands. Sei whales are seen infrequently in U.S. waters. This whale breeds and feeds in open oceans, and is generally restricted to more temperate waters. Unlike all baleen whales, the sei whale feeds mostly by filtering plankton while swimming (skim feeding), but is also known to gulp-feed krill, shrimp, and small fish.

Life History:

Sei whales become sexually mature at 6 to 12 years of age, when they reach about 45 ft (13 m) in length. They generally mate and give birth in winter, April to August in the Southern Hemisphere, November to March in the Northern Hemisphere, with gestation between 10.5 to 12.5 months in the Southern Hemisphere (Horwood 1987). Ovulation rate for sei whales in the Southern Hemisphere varies between 0.63 to 0.68 indicating a two year cycle and a true pregnancy rate of 0.41 to 0.43 (Horwood 1987; Lockyer 1974, 1984). Females breed every 2 to 3 years. They give birth to a single calf that is about 15 ft (4.6 m) long and weighs about 1,500 lbs (680 kg). Calves are usually nursed for 6 to 9 months before being weaned on the preferred feeding grounds. Sei whales have an estimated lifespan of 50 to 70 years.

Summary of Major Threats:

During the 19th and 20th centuries, sei whales were targeted and greatly depleted by commercial hunting and whaling, with an estimated 300,000 animals killed for their meat and oil. Other threats that may affect sei whale populations are ship strikes and interactions with fishing gear such as traps/pots.

Identified threats outlined in the Blue, Fin and Sei Whale Recovery Plan 2005–2010 (DEH 2005) are as follows:

- the resumption of commercial whaling and/or the expansion of scientific whaling (the IWC Convention allows member states to issue special permits to kill whales for research purposes and then process these animals for sale);
- habitat degradation due to acoustic pollution, entanglement (e.g. in marine debris, fishing and aquaculture equipment), physical injury and death from ship strike, built structures that impact upon habitat availability and/or use (e.g. marinas, wharves, aquaculture installations, mining or drilling infrastructure), changing water quality and pollution (e.g. runoff from land based agriculture, oil spills, outputs from aquaculture) and changes to water flow regimes causing extensive sedimentation or erosion or altered currents in near shore habitat (e.g. canals and dredging);
- prey depletion due to over harvesting, where sei whales rely on krill as a main food source and require adequate supplies to accumulate energy reserves essential for migration and breeding. Depletion of krill through over-harvesting may be a potential future threat for populations of these species; and
- climate and oceanographic change, potentially impacting both habitat and food availability for sei whales. Whale migration, feeding, breeding, and calving site selection may be influenced by factors such as ocean currents and water temperature. Any changes in these factors could affect recovery by rendering currently used habitat areas unsuitable. Changes to climate and oceanographic processes may also lead to decreased productivity and different patterns of prey distribution and availability. Such changes would certainly affect dependent predators such as sei whales.

4. Sperm whale, *Physeter macrocephalus*

Taxonomy:

Kingdom: Animalia
Phylum: Chordata
Class: Mammalia
Order: Cetacea
Family**:** Physeteridae
Genus: *Physeter*
Species**:** *macrocephalus*

Historical Information and Conservation-Regulatory Status:

The sperm whale was listed as endangered throughout its range on June 2, 1970 under the Endangered Species Conservation Act of 1969 (35 FR 8491). Sperm whales are also protected under the Marine Mammal Protection Act of 1972 (http://www.fisheries.noaa.gov/pr/species/mammals/whales/sperm-whale.html).

The principal cause of the decline in sperm whales was commercial whaling, and prohibitions on their harvest by the IWC have reduced the magnitude of the threat.

Species Description:

The sperm whale is the largest out of 73 different species of the toothed whale, with males reaching about up 52 ft (16 m) in length and the smaller females reaching 37 ft (12 m). Males have about 40 to 50 teeth, located only in their narrow lower jaw, while females have even fewer teeth. The sperm whale feeds primarily on squid. Plunging to 7,382 ft (2,250 m) for prey, it is the second deepest diving mammal.

This species of whales is dark gray in color, but oftentimes the interior of the mouth is bright white, and some whales have white patches on the belly. Their flippers are paddle-shaped and small compared to the size of the body, and their flukes are very triangular in shape. They have small dorsal fins that are low, thick, and usually rounded. Their skin is wrinkled to increase surface area for heat loss, giving them a shriveled look.

The sperm whale is distinguished by its extremely large head, which takes up to 25 to 35% of its total body length and is squarish in shape. It is the only living cetacean that has a single blowhole asymmetrically situated on the left side of the head near the tip. Sperm whales have the largest brain of any animal (on average 17 lbs [7.8 kg] in mature males). Atop the whale's skull is positioned a large complex of organs filled with a liquid mixture of fats and waxes called "spermaceti". The purpose of this complex is to generate powerful and focused clicking sounds, which the sperm whale uses for echolocation and communication. They use echolocation or sonar to detect objects in their environment. They produce sounds in the air passages in their heads, which are then projected out in front of them. The sound bounces off solid objects and returns to them (like an echo), so the animals are able to get a "picture" of what is around them. A lot of research is being done on whale sounds. Many species, such as the humpback and sperm whales, seem to have individually identifiable calls.

Because sperm whales spend most of their time in deep waters, their diet consists of many larger organisms that also occupy deep waters of the ocean. Their principle prey is large squid weighing between 3.5 ounces and 22 lbs (0.1 and 10 kg), but they will also eat large demersal and mesopelagic sharks, skates, and fishes. The average dive lasts about 35 minutes and is usually down 1,312 ft (400 m), however dives may last over an hour and reach depths over 3,280 ft (1,000 m).

Toothed whales tend to be social and live in groups. Most sperm whale females will form lasting bonds with other females of their family, and on average 12 females and their young will form a family unit. While females generally stay with the same unit all their lives in and around tropical waters, young males will leave when they are between 4 and 21 years old and can be found in "bachelor schools", comprising of other males that are about the same age and size. As males get older and larger, they begin to migrate to higher latitudes (toward the poles) and slowly bachelor schools become smaller, until the largest males end up alone. Large, sexually mature males that are in their late 20s or older, will occasionally return to the tropical breeding areas to mate.

Geographic Boundaries and Spatial Distribution:

As well described by NOAA, sperm whales inhabit all oceans of the world, deep waters between about 60° N and 60° S latitudes (http://www.fisheries.noaa.gov/pr/species/mammals/whales/sperm-whale.html). They can be seen close to the edge of pack ice in both hemispheres and are also common along the equator, especially in the Pacific. Their distribution is dependent on their food source and suitable conditions for breeding, and varies with the sex and age composition of the group. Sperm whale migrations are not well understood and not very predictable. In some mid-latitudes, there seems to be a general trend to migrate north and south depending on the seasons (whales move poleward in the summer). However, in tropical and temperate areas, there appears to be no obvious seasonal migration.

Sperm whales also produce a series of clicks called "codas." Each whale has a distinctive coda and scientists think that other sperm whales recognize each other by their codas. There is also evidence that they produce intense bursts of sound to stun their prey. The sperm whale is a species that is known to strand in large groups. It is not known why they strand (which means certain death), but some theories include illness, parasitic infection, following sick leaders, and malfunction of echolocation due to gently sloping beaches and underwater magnetic anomalies, which leads to disorientation.

Habitat:

Sperm whale live in every ocean in the world but stay away from the extremely cold waters near the polar ice in the north and the south. Females usually remain in temperate and tropical waters within 45-55° latitude, whereas males travel in temperate waters. Sperm whales prefer deep water around ocean trenches, where strong currents flow in opposite directions bringing concentrated nutrients to the area, and attracting a large number of creatures that the sperm whales can eat. In California, sperm whales can be seen in waters off the continental slope from November to April.

Sperm whales tend to inhabit areas with a water depth of 1968 ft (600 m) or more, and are uncommon in waters less than 984 ft (300 m) deep (http://www.fisheries.noaa.gov/pr/species/mammals/whales/sperm-whale.html). Female sperm whales are generally found in deep waters (at least 3280 ft, or 1000 m) of low latitudes (less than 40°, except in the North Pacific where they are found as high as 50°). These conditions

generally correspond to sea surface temperatures greater than 15°C, and while female sperm whales are sometimes seen near oceanic islands, they are typically far from land. According to NOAA, immature males will stay with female sperm whales in tropical and subtropical waters until they begin to slowly migrate towards the poles, anywhere between ages 4 and 21 years old. Older, larger males are generally found near the edge of pack ice in both hemispheres. On occasion, however, these males will return to the warm water breeding area.

Life History:

As reported by NOAA, male sperm whales are physically mature around 30 years and 35 feet (10.6 m) long, at which time they stop growing (http://www.fisheries.noaa.gov/pr/species/mammals/whales/sperm-whale.html). They reach sexual maturity around 9 years of age and produce a calf approximately once every five years. After a 14 to16 month gestation period, a single calf about 13 ft (4 m) long is born.

For about the first 10 years of life, male sperm whales are only slightly larger than females, but males continue to exhibit substantial growth until they are well into their 30s. They reach physical maturity around 50 years and when they are 52 ft (16 m) long. Unlike females, puberty in males is prolonged, and may last between ages 10 to 20 years old. Even though males are sexually mature at this time, they often do not actively participate in breeding until their late twenties.

Most females of the sperm whale form lasting bonds with other females of their family, and on average 12 females and their young will form a family unit. While females generally stay with the same unit all their lives in and around tropical waters, young males leave when they are between 4 and 21 years old, forming groups comprising of other males that are about the same age and size. As males get older and larger, they begin to migrate to higher latitudes (toward the poles) and slowly groups become smaller, until the largest males end up alone. Large, sexually mature males that are in their late 20s or older, will occasionally return to the tropical breeding areas to mate.

Summary of Major Threats:

Natural threats to sperm whales include killer whales, which have been documented killing at least one sperm whale in California. Typically, however, it is believed that most killer whale attacks are unsuccessful. Large sharks may also be a threat, especially for young sperm whales.

Historically (mainly between 1800 and 1987), whaling was a major threat to sperm whales, taking possibly as many as 1,000,000. Hunting of sperm whales by commercial whalers declined in the 1970s and 1980s, and virtually ceased with the implementation of a moratorium against whaling by the IWC in 1988. Currently the significant threats include ship strikes, entanglements in fishing gear and disturbance by anthropogenic noise in areas of high shipping activity and oil and gas activities In addition, the potential impact of coastal pollution may be an issue for this species in portions of its habitat, especially pollutants such as polychlorobiphenyls (PCBs), chlorinated pesticides, polycyclic aromatic hydrocarbons (PAHs), and heavy metals.

5. Humpback whale, *Megaptera novaeangliae*

Taxonomy:

Kingdom: Animalia
Phylum: Chordata
Class: Mammalia
Order: Cetacea
Family: Balaenopteridae
Genus: *Megaptera*
Species: *novaeangliae*

Historical Information and Conservation-Regulatory Status:

As well described by NOAA, in 1946, the International Convention for the Regulation of Whaling regulated commercial whaling of humpback whales (http://www.fisheries.noaa.gov/pr/species/mammals/whales/humpback-whale.html). In 1966, the IWC prohibited commercial whaling of humpbacks. In June 1970 (35 FR 8491), humpback whales were designated as "endangered" under the Endangered Species Conservation Act, which in 1973 was replaced by the ESA , where humpbacks are continued to be listed as endangered. In April 2015, NOAA proposed to revise the ESA listing of the humpback whale by identifying 14 Distinct Population Segments (DPS)s and listing 2 DPSs as threatened and 2 as endangered (80 FR 22304).

Presently, many steps are being made to recover the species of the humpback whales, which include the following efforts:

- reduce bycatch in gillnet and trap/pot fisheries in the western North Atlantic through the Atlantic Large Whale Take Reduction Plan;
- implement marine mammal take reduction measures identified in the *Pacific Offshore Cetacean Take Reduction Plan*;
- mitigate ship strikes and respond to humpback whales in distress;
- educate whale watch vessels and boat operators on practicing safe boating around whales;
- monitor humpbacks in U.S. waters via shipboard surveys and mark recapture studies; and
- research humpback population structure and abundance.

Species Description:

Humpback whales are the favorite of whale watchers, as they frequently perform aerial displays, such as breaching (jumping out of the water), or slapping the surface with their pectoral fins, tails, or heads. Scientists believe these activities are forms of communication because they create a great deal of noise, which can be heard at long distances under water. As described by NOAA, humpback whales are well known for their long "pectoral" fins, which can be up to 15 ft (4.6 m) in length and are giving them increased maneuverability used to slow down or go backwards (http://www.fisheries.noaa.gov/pr/species/mammals/whales/humpback-whale.html).

Adult females are larger than adult males, reaching lengths of up to 60 ft (18 m) and weighting 50,000 to 80,000 lbs (22,000 to 36,000 kg, 25 to 40 tons). Their body coloration is primarily dark grey, but individuals have a variable amount of white on their pectoral fins and belly. The humpback whale was given its common name because of the shape of its dorsal (back) fin and the way it looks when the animal is diving. Its scientific name, *Megaptera*, means, "large-winged" and refers to its long, white, wing-like flippers that are often as long as one-third of the animal's body length. Each whale has its own unique pattern on the underside of its tail flukes, which can be used like "fingerprints" to identify individual whales. Unique to humpbacks are round "bumps" that occur on the head forward of the blowhole and on the edges of the flippers.

Similar to other baleen whales, during the summer months, humpbacks spend the majority of their time feeding and building up fat “blubber” that they will live off of during the winter. Humpbacks filter-feed on tiny crustaceans (mostly krill), plankton, and small fish and can consume up to 3,000 lbs (1,360 kg) of food per day. Unlike other baleen whales, humpbacks can often be seen feeding cooperatively, where one or several whales blow a ring of bubbles from their blowholes that encircle a school of krill or fish. Whales then swim through the "net" with their mouths opened, taking in large amounts of food. This highly complex method of feeding, called "bubble netting," is unique to humpbacks. This technique is often performed in groups with defined roles for distracting, scaring, and herding before whales lunge at prey corralled near the surface.

Geographic Boundaries and Spatial Distribution:

Humpback whales live in all major oceans from the equator to sub-polar latitudes (http://www.fisheries.noaa.gov/pr/species/mammals/whales/humpback-whale.html). In the Southern Hemisphere, the IWC has designated seven major breeding stocks linked to seven major feeding areas. Most breeding areas for Southern Hemisphere humpbacks are at 20°S, although some are in the Northern Hemisphere, including areas along the west coast of Africa and Central America. In Costa Rica, there is overlap with Northern Hemisphere humpbacks geographically, but they are not there at the same time. All Southern Hemisphere humpbacks share feeding grounds in the Antarctic south of 40°S and between 120°E and 110°W.

Humpback whales travel great distances during their seasonal migration. In the summer, humpbacks are found in high latitude feeding grounds, such as the Gulf of Maine in the Atlantic and Gulf of Alaska in the Pacific. In the winter, they migrate to breeding grounds in subtropical or tropical waters, such as the Dominican Republic in the Atlantic and the Hawaiian Islands in the Pacific.

Habitat:

While feeding and calving, humpbacks prefer shallow waters. During calving, humpbacks are usually found in the warmest waters available at that latitude. Calving grounds are commonly near offshore reef systems, islands, or continental shores. Humpback feeding grounds are in cold, productive coastal waters. During migration, humpbacks stay near the surface of the ocean.

Life History:

In their wintering grounds, humpback whales congregate and engage in mating activities (http://www.fisheries.noaa.gov/pr/species/mammals/whales/humpback-whale.html). Breeding usually occurs once every two years, but sometimes occurs twice in a three year span. Humpbacks are generally "polygynous" with males exhibiting competitive behavior on wintering grounds. Aggressive and antagonistic behaviors include chasing, vocal and bubble displays, horizontal tail thrashing, and rear body thrashing. Males within these groups also make physical contact, striking or surfacing on top of one another.

In addition, male humpback whales sing complex songs that can last up to 20 minutes and be heard 20 miles (30 km) away. A male may sing for hours, repeating the song several times. All males in a population sing the same song, but that song continually evolves over time. Humpback whale singing has been studied for decades, but scientists still understand very little about its function.

Gestation lasts for about 11 months. Newborns are 13-16 ft (4-5 m) long and weigh about 2,000 lbs (900 kg). Weaning occurs between 6 to 10 months after birth. While mothers are protective and affectionate towards their calves, swimming close and frequently touching them with their flippers, males do not provide parental support for calves.

Summary of Major Threats:

According to NOAA (http://www.fisheries.noaa.gov/pr/species/mammals/whales/humpback-whale.html), humpback whales face presently a series of threats including the following:

- entanglement in fishing gear (bycatch) - Humpbacks can become entangled in fishing gear. NOAA has observed incidental "take" of humpback whales in the California/ Oregon swordfish and thresher shark drift gillnet fishery. Potential entanglement from gear from several fisheries can occur on their long migration from Hawaii to Alaska. Humpbacks in Hawaii have been observed entangled in longline gear, crab pots, and other non-fishery-related lines;
- ship strikes - Inadvertent ship strikes can injure or kill humpbacks. We have verified mortality related to ship strikes in the Gulf of Maine and in southeastern Alaska. Ship strikes have also been reported in Hawaii;
- whale watch harassment - Whale watching vessels may stress or even strike whales. The Gulf of Maine stock is the focus of whale watching in New England from late spring to early fall, particularly within the Stellwagen Bank National Marine Sanctuary. The central North Pacific stock is the focus of a whale-watching industry on their wintering grounds in the Hawaiian Islands. The feeding aggregation in southeast Alaska is also the focus of a developing whale-watching industry that may impact whales in localized areas; and
- habitat impacts and harvest - Shipping channels, fisheries, and aquaculture may occupy or destroy humpback whale aggregation areas. Recreational use of marine areas, including resort development and increased boat traffic, may displace whales that would normally use that area. In Hawaii, acoustic impacts from vessel operation, oceanographic research using active sonar, and military operations are also of increasing concern.

C. Species Description: Sea Turtles

1. Hawksbill sea turtle, *Eretmochelys imbricate*

Taxonomy:

Kingdom: Animalia
Phylum: Chordata
Class: Reptilia
Order: Testudines
Family: Cheloniidae
Genus: *Eretmochelys*
Species: *imbricata*

Historical Information and Conservation-Regulatory Status:

The hawksbill turtle was listed by NOAA, under ESA, as endangered species on June 2, 1970 (35 FR 8497). The agency has finalized the Recovery Plan for this species in August of 1992 (57 FR 38818) and published the Status Review on January 2, 1996 (61 FR17). On October 10, 2012, under the ESA, NOAA announced 5-year reviews of four sea turtle species including hawksbill turtles (77 FR 61573). This 5-year review was based on the best scientific and commercial data available at that time; therefore, the request was made for information on these sea turtles that has become available since their last status review in 2007. The summary of this Review was published in June of 2013 (NMFS, 2013).

Species Description:

As reported by NOAA, the hawksbill turtles are one of the smaller sea turtles. Adults are 2.5 to 3 ft in carapace (top shell) length (71 to 89 cm) and weigh between 101 and 154 lbs (46 and 70 kg) (http://www.nmfs.noaa.gov/pr/species/turtles/hawksbill.htm). Top shell of this species is elliptical in shape and bony, without ridges and has large, over-lapping scales. The top shell can be orange, brown or yellow; hatchlings are mostly brown with pale blotches on scales. Hawksbill turtles are unique among sea turtles in that they have two pairs of prefrontal scales (scales in front of its eyes) on the top of their head and each of the flippers usually has two claws.

Head of the hawksbill turtle is narrow and tapers to a point, with a beak-like mouth that gives the species its name. Their jaws are not serrated (smooth -edged). The shape of the mouth allows the hawksbill turtle to reach into holes and crevices of coral reefs to find sponges, their primary food source as adults, in addition to anemones, squid and shrimp.

Geographic Boundaries and Spatial Distribution:

According to NOAA, hawksbill turtles are most tropical of all sea turtles (http://www.nmfs.noaa.gov/pr/species/turtles/hawksbill.htm). They can be found from 30° N to

30° S latitude in the Atlantic, Pacific, and Indian Oceans. Hawksbills are widely distributed throughout the Caribbean Sea and western Atlantic Ocean, regularly occurring in southern Florida and the Gulf of Mexico, in the Greater and Lesser Antilles, and along the Central American mainland south to Brazil.

In addition to nesting beaches in the U.S. Caribbean, hawksbills nest at numerous other sites throughout the Caribbean, with the majority of nesting occurring in Mexico and Cuba. The largest nesting population of hawksbills appears to occur in Australia.

Within the U.S., hawksbills are most common in Puerto Rico and its associated islands and in the Virgin Islands. In the continental U.S., hawksbills are found primarily in Florida and Texas, though they have been recorded in all the Gulf States and along the east coast as far north as Massachusetts.

The most significant nesting within the U.S. occurs in Puerto Rico and the Virgin Islands, specifically on Mona Island and Buck Island, respectively. Nesting also occurs on other beaches in St. Croix and on St. John, St. Thomas, Culebra Island, Vieques Island, and mainland Puerto Rico. Within the continental U.S., nesting is restricted to the southeast coast of Florida and the Florida Keys, but nesting is rare in these areas. No nesting occurs on the west coast of the U.S. mainland.

Habitat:

Hawksbill turtles are typically found around coastal reefs, rocky areas, estuaries and lagoons. According to NOAA, hawksbill turtles use different habitats at different stages of their life cycle, but are most commonly associated with healthy coral reefs (http://www.nmfs.noaa.gov/pr/species/turtles/hawksbill.htm).

In the Atlantic, juveniles of this species are believed to occupy the pelagic environment, taking shelter in floating algal mats and drift lines of marine debris. In the Pacific, the open-ocean (pelagic) habitat of hawksbill juveniles is unknown. After a few years in the pelagic zone, small juveniles recruit to coastal "nonbreading" areas (foraging grounds). This shift in habitat also involves a shift in feeding strategies, from feeding primarily at the surface to feeding below the surface primarily on animals associated with coral reef environments.

The ledges and caves of coral reefs provide shelter for resting hawksbills both during the day and at night. Hawksbills are known to inhabit the same resting spot night after night. Hawksbills are also found around rocky outcrops and high energy shoals, which are also optimum sites for sponge growth. They are also known to inhabit mangrove-fringed bays and estuaries, particularly along the eastern shore of continents where coral reefs are absent.

In 1998, NOAA designated critical habitat for hawksbill turtles in coastal waters surrounding Mona and Monito Islands, Puerto Rico (63 FR46693).

Life History:

As reported by NOAA, female hawksbills return to the beaches where they were born (natal beaches) every 2 to 4 years to nest (http://www.nmfs.noaa.gov/pr/species/turtles/hawksbill.htm).

Research indicates that adult hawksbill turtles are capable of migrating long distances between nesting beaches and foraging areas, which are comparable to migrations of green and loggerhead turtles. They usually nest high up on the beach under or in the beach/dune vegetation. They commonly nest on pocket beaches, with little or no sand. They nest at night, and they nest about every 14-16 days during the nesting season. The nesting season varies with locality, but in most locations nesting occurs sometime between April and November. Hawksbill turtles nest between 3 to 6 times per season, at intervals of 2 to 4 years. In each nest, they lay an average 130 eggs which incubate for about 60 days.

Male hawksbills mature when they are about 27 in (70 cm) long. Females mature at about 30 in (80 cm). The ages at which turtles reach these lengths are unknown.

Summary of Major Threats:

Hawksbills turtles face threats on nesting beaches and in the marine environment. The primary global threat to hawksbills is habitat loss of coral reef communities caused by human activities due to gradually released pollution or toxic spills and vessel groundings. In addition, global climate change is negatively impacting coral reefs by causing higher incidences of coral diseases, which can ultimately kill entire coral reef communities. Hawksbill turtles rely on coral reefs for food resources and habitat. As these communities continue to decline in quantity and quality, hawksbills will have reduced foraging opportunities and limited habitat options. Incidental capture in fishing gear, primarily gillnets, and vessel strikes also adversely affect this species' recovery.

Historically, commercial exploitation was the primary cause of the decline of hawksbill sea turtles. There remains a continuing demand for the hawksbill's shell as well as other products, including leather, oil, perfume, and cosmetics. Additionally, hawksbills are harvested for their eggs and meat. Increased recreational and commercial use of nesting beaches, beach camping and fires, litter and other refuse, general harassment of turtles, and loss of nesting habitat from human activities also negatively impact hawksbills.

2. Green sea turtle, *Chelonia mydas*

Taxonomy:

Kingdom: Animalia
Phylum: Chordata
Class: Reptilia
Order: Testudines
Family: Cheloniidae
Genus: *Chelonia*
Species: *mydas*

Historical Information and Conservation-Regulatory Status:

The green turtle was listed by NOAA and FWS as endangered species, under the ESA, on July 28, 1978 (43 FR 32800). The breeding populations in Florida and the Pacific coast of Mexico are listed as endangered; elsewhere the species is listed as threatened.

In March 2015, NOAA and FWS proposed to remove the current range-wide listing of the green sea turtle and, in its place, to list 8 DPSs as threatened and 3 DPSs as endangered (80 FR 15271). NOAA has extended the public comment period for this proposal through September 25, 2015.

Species Description:

Green turtles are easily distinguished from other sea turtles because they have a single pair of prefrontal scales (scales in front of its eyes), rather than two pairs as found on other sea turtles. Their head is small and blunt with a serrated (tooth-edged) jaw (http://www.nmfs.noaa.gov/pr/species/turtles/green.htm). Adult green turtles are unique among sea turtles in that, as herbivorous, they eat only plants and are feeding primarily on seagrasses and algae. This diet is thought to give them greenish-colored fat, from which they take their name.

Green turtles are the largest of all the hard-shelled sea turtles, but have a comparatively small head (Sea Turtles Conservancy website: http://www.conserveturtles.org/seaturtleinformation). Adults often reach 3 to 4 ft (83 to 114 cm) in the length of carapace and weigh between 240 and 420 lbs (110 to 190 kg). The carapace is bony without ridges and has large, non-overlapping, scutes (scales) present with only 4 lateral scutes. Their body is nearly oval. All flippers have one visible claw.

The carapace color varies from pale to very dark green and plain to very brilliant yellow, brown and green tones with radiating stripes. The bottom shell (plastron) varies from white, dirty white or yellowish in the Atlantic populations to dark grey-bluish-green in the Pacific populations. Hatchlings are dark-brown or nearly black with a white underneath and white flipper margins.

Geographic Boundaries and Spatial Distribution:

The green turtle is globally distributed and generally found in tropical and subtropical waters along continental coasts and islands between 30° North and 30° South (Sea Turtles Conservancy website: http://www.conserveturtles.org/seaturtleinformation). Green turtles are thought to inhabit coastal areas of more than 140 countries, where the nesting occurs in over 80 countries throughout the year.

In U.S. Atlantic and Gulf of Mexico waters, green turtles are found in inshore and nearshore waters from Texas to Massachusetts, the U.S. Virgin Islands, and Puerto Rico. Important feeding areas in Florida include the Indian River Lagoon, the Florida Keys, Florida Bay, Homosassa, Crystal River, Cedar Key, and St. Joseph Bay.

In the eastern North Pacific, green turtles have been sighted from Baja California to southern Alaska, but most commonly occur from San Diego south. In the central Pacific, green turtles

occur around most tropical islands, including the Hawaiian Islands. Adult green turtles that feed throughout the main Hawaiian Islands undergo a long migration to French Frigate Shoals in the Northwest Hawaiian Islands, where the majority of nesting and mating occurs.

Habitat:

In 1998, NOAA designated critical habitat for green turtles in coastal waters surrounding Mona and Monito Islands, Puerto Rico (63 FR46693). According to this Federal Register notice, green turtles are rarely observed in the open ocean and mainly stay near the coastline and around islands, living in bays and protected shores with seagrass beds.

Green turtles primarily use three types of habitat: beaches for nesting, open ocean convergence zones and the coastal areas for benthic feeding (Sea Turtles Conservancy website: http://www.conserveturtles.org/seaturtleinformation). Adult females migrate to mainland or island nesting beaches, traveling hundreds or thousands of kilometers each way. After emerging from the nest, hatchlings swim to offshore areas, where they are believed to live for several years, feeding close to the surface on a variety of pelagic plants and animals. Once the juveniles reach a certain age and size range, they leave the pelagic habitat and travel to nearshore benthic habitats, feeding on sea grasses and algae.

Life History:

While nesting season varies from location to location in the southeastern U.S., females of the green turtles species generally nest in the summer between June and September with the peak nesting occurring in June and July. During the nesting season, females nest at approximately two-week intervals. Green turtles nest between 3 to 5 times per season, at intervals of about every 2 years, with wide year-to-year fluctuations in numbers of nesting females. Each female lays an average 115 eggs in each nest, with the eggs incubating for about 60 days.

Summary of Major Threats:

The principal cause of the historical, worldwide decline of the green turtle is long-term harvest of eggs and adults on nesting beaches and juveniles and adults on feeding grounds (Sea Turtles Conservancy website: http://www.conserveturtles.org/seaturtleinformation.php). These harvests continue in some areas of the world and compromise efforts to recover this species.

Incidental capture in fishing gear, primarily in gillnets, but also in trawls, traps and pots, longlines, and dredges is also a serious cause of mortality that adversely affects the species recovery. Overall, green turtles are subject to the same threats as other marine turtles including habitat degradation caused by human activities due to gradually released pollution or toxic spills and vessel groundings. In addition, green turtles are also threatened, in some areas of the world, by a disease known as fibropapillomatosis.

3. Loggerhead sea turtle, *Caretta caretta*

Taxonomy:

Kingdom: Animalia
Phylum: Chordata
Class: Reptilia
Order: Testudines
Family: Cheloniidae
Genus: *Caretta*
Species: *caretta*

Historical Information and Conservation-Regulatory Status:

The loggerhead turtle was first listed by NOAA and FWS, under the ESA as threatened throughout its range on July 28, 1978 (43 FR 32800). The original Recovery Plan for this species was published by the NOAA and FWS in September 1984 and the first revision was completed in December 1991 (http://www.nmfs.noaa.gov/pr/species/turtles/loggerhead.htm). In December 2008, NOAA and FWS finalized the second revision of the Recovery Plan for loggerheads in the U.S. Caribbean, Atlantic, and Gulf of Mexico. In 2009, both agencies published an updated status review and, in 2010, proposed to list nine DPSs of loggerhead sea turtles, under the ESA. All of nine DPS were listed in September of 2011.

Species Description:

Loggerheads were named for their relatively large heads, which support powerful jaws and enable them to feed on hard-shelled prey, such as whelks and conch (Sea Turtles Conservancy website: http://www.nmfs.noaa.gov/pr/species/turtles/loggerhead.htm). Loggerheads can reach 3 ft in length and weigh 250 lbs (113 kg).

The top shell (carapace) is slightly heart-shaped and reddish-brown in adults and sub-adults, while the bottom shell (plastron) is generally a pale yellowish color. The neck and flippers are usually dull brown to reddish brown on top and medium to pale yellow on the sides and bottom. Hatchlings lack the reddish-brown coloration of adults and juveniles. Their flippers are dark gray to brown above with white to white-gray margins. The coloration of the plastron is generally yellowish to tan.

Although they are good swimmers, loggerheads have callus-like traction scales beneath their flippers that allow them to "walk" on the ocean floor (http://www.nmfs.noaa.gov/pr/species/turtles/loggerhead.htm.).

Geographic Boundaries and Spatial Distribution:

As described by NOAA, loggerheads are occurring throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans (http://www.nmfs.noaa.gov/pr/species/turtles/loggerhead.htm). Loggerheads are the most abundant species of sea turtle found in U.S. coastal waters.

In the Atlantic, the loggerhead turtles range from Newfoundland to Argentina. During non-nesting years, adult females from U.S. beaches are distributed in waters off the eastern U.S. and throughout the Gulf of Mexico, Bahamas, Greater Antilles, and Yucatán. During the summer, nesting occurs primarily in the subtropics with the major nesting concentrations in the U.S. found from North Carolina through southwest Florida, and a minimal nesting occurring westward to Texas and northward to Virginia. The majority of loggerhead nesting occurs in the western rims of the Atlantic and Indian Oceans. The loggerhead nesting aggregations in Oman, the U.S., and Australia account for about 88 % of nesting worldwide. In the southeastern U.S., about 80 % of loggerhead nesting occurs in six Florida counties.

In the eastern Pacific, loggerheads have been reported from Alaska to Chile. In the U.S., occasional sightings are reported from the coasts of Washington and Oregon, but most records are of juveniles off the coast of California. The west coast of Mexico, including the Baja Peninsula, provides critically important developmental habitats for juvenile loggerheads. The only known nesting areas for loggerheads in the North Pacific are found in southern Japan.

Habitat:

On July 10, 2014, both NOAA and FWS designated critical habitat for the Northwest Atlantic DPSs for loggerhead sea turtles in waters and beach habitat of the Gulf of Mexico and along the coast of the U.S. Atlantic Ocean (79 FR 39855).

Loggerheads occupy three different ecosystems during their lives: ocean beaches (terrestrial zone), water (oceanic zone) and nearshore coastal areas (neritic zone). Loggerheads nest on ocean beaches, generally preferring high energy, relatively narrow, steeply sloped, coarse-grained beaches. Immediately after hatchlings emerge from the nest, they begin an active period and move from their nest to the surf, swim, and are swept through the surf zone, and continue swimming away from land for up to several days. After this active period, post-hatchling loggerheads take up residence in areas where surface waters converge to form local downwellings (Witherington, 2002). As post-hatchlings, loggerheads may linger for months in waters just off the nesting beach or become transported by ocean currents within the Gulf of Mexico and North Atlantic. Once individuals get transported by ocean currents farther offshore, they've entered the oceanic zone (Bolton 2003). Somewhere between 7 and 12 years old, oceanic juveniles migrate to nearshore coastal areas (neritic zone) and continue maturing until adulthood.

In addition to providing critically important habitat for juveniles, the neritic zone also provides crucial foraging habitat, inter-nesting habitat, and migratory habitat for adult loggerheads in the western North Atlantic. To a large extent, these habitats overlap with the juvenile stage, the exception being most of the bays, sounds, and estuaries along the Atlantic and Gulf coasts of the U.S. from Massachusetts to Texas, which are infrequently used by adults. However, adult loggerheads are present year-round in Florida Bay, an important feeding area, probably because of relatively easy access to open ocean and migratory routes.

The predominate foraging areas for western North Atlantic adult loggerheads are found throughout the relatively shallow continental shelf waters of the U.S., Bahamas, Cuba, and the

Yucatán Peninsula, Mexico. Seasonal migrations of adult loggerheads along the mid- and southeast U.S. coasts have also been documented.

Life History:

According to the National Wildlife Federation, female loggerheads reach maturity at about 35 years of age (https://www.nwf.org/Wildlife/Wildlife-Library/Amphibians-Reptiles-and-Fish/Sea-Turtles/Loggerhead-Sea-Turtle.aspx). Every 2to 3 years, loggerheads mate in coastal waters and then return to nest on the very same beach where they were hatched, called the "natal" beach. The nesting season begins in April and ends in September, with the peak in June. They emerge onto the beach at night every 14 days, laying an average of 4 clutches containing roughly 100 to 120 eggs in each. The eggs incubate approximately two months before hatching. Sex of hatchlings is determined by incubation temperature, where warmer temperatures result in the great majority being females and cooler temperatures produce mainly or only males.

According to NOAA, during the 3 months or so that a female loggerhead breeds, she will travel hundreds of miles to nest, lay 35 lbs (16 kg) of eggs (or more) and swim back to her home foraging (non-nesting) area, all without eating anything significant (http://www.nmfs.noaa.gov/pr/species/turtles/loggerhead.htm).

Summary of Major Threats:

Loggerhead turtles face threats on both nesting beaches and in the marine environment. According to NOAA (http://www.nmfs.noaa.gov/pr/species/turtles/loggerhead.htm), major threats include: degradation or loss of nesting and foraging habitats; disorientation of hatchlings by beachfront lighting; excessive nest predation by native and non-native predators; marine pollution (including oil spills and debris); vessels strikes; disease; incidental take from channel dredging and commercial trawling, longline, and gill net fisheries as well as directed harvest.

4. Leatherback sea turtle, *Dermochelys coriacea*

Taxonomy:

Kingdom: Animalia
Phylum: Chordata
Class: Reptilia
Order: Testudines
Family: Dermochelyidae
Genus: *Dermochelys*
Species: *coriacea*

Historical Information and Conservation-Regulatory Status:

The leather turtle was listed by NOAA, under ESA, as endangered species on June 2, 1970 (35 FR 8491). In 1991, both NOAA and FWS published the Recovery Plan for this species. The original 5-year review for the species was finalized in 2007. On October 10, 2012, under the

ESA, NOAA announced 5-year reviews of four sea turtle species, including leatherback (*Dermochelys coriacea*) sea turtles (77 FR 61573). This 5-year review was based on the best scientific and commercial data available at that time; therefore, the request was made for information on these sea turtles that has become available since their last status review in 2007. This review was published in November 2013.

Species Description:

According to NOAA, the leatherback is the largest turtle and one of the largest living reptiles in the world (http://www.nmfs.noaa.gov/pr/species/turtles/leatherback.htm). This species can reach up to 6.6 ft (2m) and weigh up to 2,000 lbs (900 kg). The leatherback is the only sea turtle that doesn't have a hard bony shell. Leatherback turtles are named for their shell, which is leather-like rather than hard, like other turtles. A leatherback's carapace is about 1.5 in (4 cm) thick and consists of leathery, oil-saturated connective tissue overlaying loosely interlocking dermal bones.

Their front flippers don't have claws or scales and are proportionally longer than in other sea turtles. Their back flippers are paddle-shaped. Both their ridged carapace and their large flippers make the leatherback uniquely equipped for long distance foraging migrations.

Leatherbacks don't have the crushing chewing plates, characteristic of other sea turtles that feed on hard-bodied prey (Pritchard 1971). Instead, they have pointed tooth-like cusps and sharp-edged jaws that are perfectly adapted for a diet of soft-bodied pelagic (open ocean) prey, such as jellyfish and salps. A leatherback's mouth and throat also have backward-pointing spines that help retain such gelatinous prey. Leatherbacks can dive to depths of 4,200 ft (1,280 m), deeper than any other turtle, and can stay down for up to 85 minutes.

Thermoregulatory adaptations such as a counter-current heat exchange system, high oil content, and large body size allow them to maintain a core body temperature higher than that of the surrounding water, thereby allowing them to tolerate colder water temperatures.

Geographic Boundaries and Spatial Distribution:

Leatherbacks have the widest global distribution of all reptile species. The leatherback turtle is distributed worldwide in tropical and temperate waters of the Atlantic, Pacific, and Indian Oceans. It is also found in small numbers as far north as British Columbia, Newfoundland, and the British Isles, and as far south as Australia, Cape of Good Hope, and Argentina.

As reported by NOAA, the global population of leatherbacks comprises seven biologically and geographically subpopulations, which are located in the Atlantic, Pacific, and Indian Ocean (NMFS 2009b).

The subpopulations with ranges overlapping U.S. territory are the West Pacific, East Pacific, and Northwest Atlantic leatherbacks. Western Pacific leatherbacks feed off the Pacific Coast of North America, and migrate across the Pacific to nest in Malaysia, Indonesia, Papua New Guinea, and the Solomon Islands. Eastern Pacific leatherbacks, on the other hand, nest along the Pacific coast of the Americas in Mexico and Costa Rica.

Leatherback turtle nesting grounds are located around the world. The largest remaining nesting assemblages are found on the coasts of Northern South America and West Africa. In addition, nesting colonies are observed in Puerto Rico, the USVI and Southeast Florida. The distribution and developmental habitats of juvenile leatherbacks are poorly understood. Adult leatherbacks are capable of tolerating a wide range of water temperatures and have been sighted along the entire continental east coast of the U.S. as far north as the Gulf of Maine and south to Puerto Rico, the USVI, and into the Gulf of Mexico.

Habitat:

Leatherbacks are commonly known as pelagic (open ocean) animals, but they also forage in coastal waters. In fact, leatherbacks are the most migratory and wide ranging of sea turtle species. Leatherbacks mate in the waters adjacent to nesting beaches and along migratory corridors. After nesting, female leatherbacks migrate from tropical waters to more temperate latitudes, which support high densities of jellyfish prey in the summer.

Pursuant to a joint NOAA-NMFS and FWS agreement, the FWS has jurisdiction over sea turtles on the land and NOAA over sea turtles in the marine environment. The FWS initially designated critical habitat for leatherback turtles on September 26, 1978 (43 FR 12050). The critical habitat area consisted of a strip of land 0.2 miles wide (from mean high tide inland) in the USVI at Sandy Point Beach on the western end of the island of St. Croix. In 1979, NOAA has designated critical habitat for leatherback turtles to include the coastal waters adjacent to Sandy Point, St. Croix, USVI (44 FR 17710). NOAA designated critical habitat to provide protection for endangered leatherback sea turtles along the U.S. West Coast in January 2012 (77 FR 4170).

Life History:

The leatherback life cycle is broken into several stages: (1) egg/hatchling; (2) post-hatchling; (3) juvenile; (4) sub-adult; and (5) adult. There is still uncertainty regarding the age at first reproduction. Female leatherbacks typically nest on sandy, tropical beaches at intervals of 2 to 4 years (McDonald and Dutton 1996). Females lay clutches of approximately 100 eggs several times during a nesting season, typically at 8-12 day intervals. After about 2 months, leatherback hatchlings emerge from the nest and have white striping along the ridges of their backs and on the margins of the flippers.

The data suggest that leatherbacks follow a life history strategy similar to many other long-lived species that delay age of maturity, have low and variable survival in the egg and juvenile stages, and have relatively high and constant annual survival in the sub-adult and adult life stages (Chaloupka 2002).

Summary of Major Threats:

Leatherback turtles face threats on both nesting beaches and in the marine environment. The greatest causes of decline and the continuing primary threats to leatherbacks worldwide are degradation of habitat due to human activities, pollution of water, long-term harvest and incidental capture in fishing gear. Harvest of eggs and adults occurs on nesting beaches while juveniles and adults are harvested on feeding grounds. Incidental capture primarily occurs in

gillnets, but also in trawls, traps and pots, longlines, and dredges. Together these threats are serious ongoing sources of mortality that adversely affect the species' recovery.

D. Species Description: Fish

1. Scalloped Hammerhead Shark, *Sphyrna lewini*

Taxonomy:

Kingdom: Animalia
Phylum: Chordata
Class: Chondrichthyes
Order: Carcharhiniformes
Family: Sphyrnidae
Genus: *Sphyrna*
Species: *lewini*

Historical Information and Conservation-Regulatory Status:

On August 14, 2011, NOAA received a petition from Wild Earth Guardians and Friends of Animals to list the scalloped hammerhead shark as threatened or endangered under the ESA throughout its entire range, or, as an alternative, to delineate the species into DPSs. On November 28, 2011, NOAA published a notice that listing may be warranted and published a proposed rule to list scalloped hammerhead shark under the ESA and the status review of the species in April 2013. In July 2014, the Agency listed 4 DPSs under the ESA.

Species Description:

According to NOAA, scalloped hammerhead sharks are moderately large sharks with a global distribution (http://www.fisheries.noaa.gov/pr/species/fish/scalloped-hammerhead-shark.html). The adult species can reach 5 to 11 ft in length and 335 lbs (152 kg).

Like all representatives of this family, scalloped hammerhead has the typically formed "hammer" on its head with eyes and nostrils located at the tips of the extensions. The flat, extended head (cephalofoil) of a scalloped hammerhead shark is characterized by an indentation located centrally on the front margin of the broadly arched head. Two more indentations flank the main central indentation, giving this hammerhead a "scalloped" appearance.

Another typical characteristic of this species is the free end tip of the second dorsal fin which almost reaches the tail fin. Their coloring is mainly olive, bronze or light brown with a white belly. The edges of the fins are usually darker on young animals, but become lighter as they grow older.

This hammerhead species feeds mostly on fish such as sardines, herring and mackerels, occasionally also on invertebrates such as octopuses. Large scalloped hammerhead sharks also eat small-sized shark species such as the Atlantic sharpnose shark or the blacktip reef shark. They have a very high metabolic rate, thus young sharks need a significant amounts of food, or risk starving to death.

Geographic Boundaries and Spatial Distribution:

The scalloped hammerhead is a coastal, pelagic, species, it occurs over continental and insular shelves and in nearby deeper water. According to NOAA, scalloped hammerhead sharks are found worldwide residing in coastal warm temperate and tropical seas in the Atlantic, Pacific, and Indian Oceans between 46°N and 36°S. During the day scalloped hammerheads are more often found close to shore and at night they hunt further offshore.

Habitat:

Adults occur alone, in pairs or in small schools (Hazin et al. 2001). Young sharks tend to form huge schools whose function is presumed to be not only feeding and reproduction, but also protection (these animals have practically no natural enemies after reaching full maturity). Groups of scalloped hammerheads prefer staying in regions which have seamounts reaching from great depths practically to the water's surface. Latest research also shows that these sharks can make use of the earth's magnetic field during their migrations.

They move in the night and use the environment as a map. These sharks utilize a point to point type of school swimming, and do not favor going too deep where temperature changes are impacted by current speed and directional change. The scalloped hammerhead utilizes deep-water to survive as safety and feeding. Although they have high metabolic rates, they have a tendency to be sedentary and allow currents to carry them as they swim. As a result, this causes the scalloped hammerhead to be selective where they swim and the depth at which they tend to stay at.

Life History:

Males and females of the scalloped hammerheads differ in that, in general, male sharks are observed to stay deeper in the ocean waters than female sharks (http://www.fisheries.noaa.gov/pr/species/fish/scalloped-hammerhead-shark.html). Mature females can reach a length of 4 m or more, although the average length is less. Males reach sexual maturity at a length of about 1.6 m and females when they reach approximately 2.1m. The pups measure approximately 0.5 m, at birth. Young scalloped hammerheads grow relatively slowly when compared to other shark species.

Research suggests that females are capable of giving birth annually, usually in the summer. The gestation period for scalloped hammerhead shark is reported to be around 12 months (Hazin et al. 2001). Scalloped hammerheads bear their young alive. Pregnancy lasts between 9 and 10 months. Compared to other species, this species produces large litters, which is most likely due to high infant mortality. Depending on their size, the females give birth to between 15 and

30 pups. The "hammer" is made of cartilage and is very soft when the young are born so as to ease the birth process.

Like most sharks, parental care is not observed. Nursery grounds for this species are predictable and repeated over the years, and it is recorded that they are very faithful to their natal sites.

Summary of Major Threats:

The scalloped hammerhead shark species is highly desired for the shark fin trade because of its fin size and high fin ray count. They are caught in a variety of fisheries including artisanal and small-scale commercial fisheries, bottom longlines as well as offshore pelagic longlines, gillnets, etc. They are valuable in the international fin market and are often used to make shark fin soup.

2. Nassau grouper, *Epinephelus striatus*

Taxonomy:

Kingdom: Animalia
Phylum**:** Chordata
Class: Actinopterygii
Order: Perciformes
Family**:** Serranidae
Genus: *Epinephelus*
Species: *striatus*

Historical Information and Conservation-Regulatory Status:

There has been a complete ban on the fishing of Nassau grouper in the U.S. federal waters since 1990. This includes federal waters around Puerto Rico and the USVI. There is also a ban on the fishing of Nassau grouper in state waters within the U.S., as well. The species is a candidate for the U.S. Endangered Species List.

The NOAA-NMFS has designated the Nassau grouper as “species of concern” (NMFS 2010a). On August 31, 2010, a petition was submitted, by Wild Earth Guardians, to NOAA and FWS to list Nassau grouper under the ESA. In September 2014, NOAA published a “Proposed Rule: Notice of 12-Month Finding on Petition to list under the ESA” (79 FR 51929). The final determination is expected to be published by NOAA and FWS within 12 months of the proposed rule publication.

Species Description:

According to NOAA, Nassau grouper species are reaching a maximum size of 122cm (48in) in length and maximum weight of 25 kg (NMFS 2010a). The Nassau grouper is one of the larger serranids of the tropical Western Atlantic and Caribbean. Nassau grouper is a slow-growing and long-lived species, with estimated life span of three decades.

This species is characterized by five dark brown vertical bars on pale tan or gray body, black dots around the eye, a large black saddle-blotch on the caudal peduncle (the narrow part of the body to which the tail attaches), and a wide "tuning fork" pattern on their forehead. However, they can greatly lighten or darken their overall pattern within minutes.

The Nassau Grouper is a top-level predator whose diet consists mainly of fish and crabs. They are ambush suction foragers, which lie and wait for prey and then entirely engulf the organism.

Geographic Boundaries and Spatial Distribution:

The Nassau Grouper is found from Bermuda and Florida throughout the Bahamas and Caribbean Sea, including the Gulf of Mexico and up the Atlantic coast to North Carolina (NOAA 2010a). The NMFS defines the Species of Concern range for the Nassau Grouper as including the U.S. Atlantic Coast and Gulf of Mexico.

The Nassau grouper is, primarily, a shallow-water, insular species that has long been valued as a major fishery resource throughout the wider Caribbean, South Florida, Bermuda and the Bahamas.

The Nassau grouper is listed as "Native" to the following countries/states: Anguilla; Antigua and Barbuda; Aruba; Bahamas; Barbados; Belize; Bermuda; Cayman Islands; Colombia; Costa Rica; Cuba; Curaçao; Dominica; Dominican Republic; French Guiana; Grenada; Guadeloupe; Guatemala; Guyana; Haiti; Honduras; Jamaica; Mexico; Montserrat; Netherlands Antilles (Curaçao); Nicaragua; Panama; Puerto Rico; Saint Kitts and Nevis; Saint Lucia; Saint Vincent and the Grenadines; Suriname; Trinidad and Tobago; Turks and Caicos Islands; United States (Florida); United States Minor Outlying Islands (Caribbean: i.e., Navassa Island); Venezuela; British Virgin Islands and the USVI.

Habitat:

The Nassau grouper is considered a reef fish, but it transitions through a series of ontogenetic shifts of both habitat and diet (WildEarth Guardians 2010). As larvae, they are found in planktonic waters, where as juveniles they are found in nearshore shallow waters in macroalgal and seagrass habitats. They shift progressively deeper with increasing size and maturation into predominantly reef habitat. Larger adults tend to occupy deeper, more rugose, reef areas. Adult Nassau grouper tend to be relatively sedentary and are found most abundantly on high relief coral reefs or rocky substrate in clear waters, although they can be found from the shoreline to about 100 to 130 m.

Life History:

Male and female Nassau grouper typically atteins sexual maturity by about 50 cm and about 4 to 5 years of age (WildEarth Guardians 2010). As with most large marine reef fishes, Nassau grouper demonstrate a bi-partite life cycle with demersal juveniles and adults (leaving near the bottom), but pelagic eggs and larvae (existing in the open waters). Reproduction is only known to occur during annual aggregations, in which large numbers of Nassau grouper, ranging from dozens to tens of thousands, collectively spawn (Colin 1992). Many fish travel long distances to arrive at predictable places during the few weeks, spread over several months, each year when

spawning occurs and then return to their home reefs (Sadovy and Eklund 1999). Fertilization is external. Fertilized eggs hatch after 23 to 40 hours depending on environmental temperatures. After hatching, pelagic larval duration may range from 42 to 70 days with transformation from pelagic to demersal form occurring in less than one week.

Summary of Major Threats:

There are several sources of threat to the Nassau grouper identified in the literature (WildEarth Guardians 2010). The primary threat to these grouper is overfishing, whether intentionally or as bycatch, from gill-nets, long-lines, bottom trawls, and other fishing activities. Fishing had been identified as a primary source of destruction to this species, where two different aspects of fishing effect Nassau grouper stocks, fishing effort throughout the non-spawning months and fishing effort directed at spawning aggregations or migratory access to spawning aggregations

An underlying driver of the fishing threat is high human population density and growth, which is likely to cause more habitat destruction, including more energy development and specifically near and offshore oil drilling, which could have a devastating impact on the habitats of these grouper. As reported by NOAA, habitat loss or degradation is another significant threat to this species (NMFS 2010a). As previously described, during its various life history stages, the Nassau grouper uses many different communities or habitat types within the coral reef ecosystem. The increase in urban, industrial, and tourist developments throughout the species' range impacts coastal mangroves, seagrass beds, estuaries, and live coral (Mahon 1990). Loss of juvenile habitat, such as macroalgae, seagrass beds, and mangrove channels is likely to negatively affect recruitment rates. Poor water quality is a threat to both corals and macroalgae in nearshore areas. Increased sedimentation resulting from poor land development practices adds turbidity and pollutants into nearshore habitats and can change water flow patterns in creeks, where newly settled juveniles may be found. Dredging operations are also capable of destroying macroalgal beds that may be used as grouper nursery areas.

Climate change also has its implication on Nassau grouper survival (NMFS 2010a). This species have been found across a range of temperatures, however spawning occurs only when sea surface temperatures are approximately 25°C. If sea surface temperatures rise, the geographic range of the species may shift. One of the other potential effects of climate change could relate to the loss of structural habitat in the coral reef ecosystems (Munday et al. 2008). Increased sea surface temperatures have been responsible for coral loss through bleaching and disease reducing adult habitat for Nassau grouper (Coleman and Koenig 2010). In addition, ocean acidification is anticipated to affect the integrity of coral reefs and changing sea level could modify the depth regime with such rapidity that coral and coral reefs will be affected (Munday et al. 2008).

VII. The USEPA activities related to coral reef protection

A. U.S. EPA - Efforts on National Level

1. The U.S. Coral Reef Task Force

The U.S. Coral Reef Task Force (USCRTF) was established in 1998 by Presidential Executive Order 13089, which mandates that federal agencies use their programs and authorities to protect and enhance U.S. coral reef ecosystems and ensure that any authorized, funded, or executed action will not degrade the conditions of these ecosystems (Maurin and Bobbe 2009; National Ocean Service website: http://www.coralreef.gov/). The Task Force currently consists of 12 Federal agencies (NOAA, Department of Interior (DOI), the U.S. Agency for International Development (USAID), Department of Agriculture, Department of Defense, the U.S. Coast Guard, Department of Justice, Depatment of Statistics, Department of Transportation, National Aeronautics and Space Administration, Antional Science Foundation (NSF), and EPA); seven U.S. states, territories, commonwealths (the Commonwealth of the Northern Mariana Islands, Puerto Rico, Florida, Hawaii, Guam, American Samoa, and USVI); and the three U.S. Freely Associated States (the Federated States of Micronesia, the Republic of the Marshall Islands and the Republic of Palau). EPA's designee to the USCRTF is the Assistant Administrator for Water (http://water.epa.gov/type/oceb/habitat/taskforce.cfm). EPA has taken a strong role in protecting coral reefs in the USCRTF jurisdictions through research, grant funding, technical assistance, and program development, implementation, and enforcement. EPA has focused its efforts both nationally and regionally on addressing the threats to coral reefs from land-based sources of pollution. There are numerous offices within EPA that address coral reef protection: Office of Water, Office of Research and Development (ORD), Office of Environmental Information, and several EPA regional offices.

EPA is an active participant in the USCRTF (http://water.epa.gov/type/oceb/habitat/taskforce). Region 2 is one of three EPA's Regional Offices represented on the Task. Region 2 has the responsibility for working with Puerto Rico and the USVI's jurisdictions. Coral reef-related programs include the following:

- EPA's Environmental Monitoring and Assessment Program;
- National Coastal Condition Report;
- EPA's Marine Debris Prevention Program;
- Biocriteria for the Protection of Coral Reefs; and
- Ecological Research Program.

2. The United States Coral Reef Initiative

In 1996, the United States launched the United States Coral Reef Initiative (USCRI), which has been created to support national and international coral reef conservation efforts (http://water.epa.gov/type/oceb/habitat/initiative.cfm). The USCRI consists of federal, state, territorial and commonwealth governments, the nation's scientific community, the private sector, and other organizations. NOAA is one of the prime federal agency contributors to the USCRI along with the DOI, NSF, USAID and EPA. It is supported not only in the U.S., but also in Japan, Australia, and Jamaica. The primary goal is to strengthen and fill the gaps in existing efforts to conserve and sustainably manage coral reefs and related ecosystems (sea grass beds and mangrove forests) in U.S. waters.

B. EPA Region 2 – Efforts on Regional Level

1. Caribbean Coral Reef Partnership

In February, 2013, EPA Region 2 initiated an inter-agency partnership to protect coral reefs off the shores of Puerto Rico and the USVI. The Caribbean Coral Reef Protection Group, which consists of the 13 federal and local agencies, was formed to facilitate a closer working relationship among its member agencies to coordinate more effective government strategies in protecting coral reefs in the Caribbean.

In 2014, EPA reorganized the Coral Reef Protection Group to become the Caribbean Coral Reef Partnership (CCRP), and established co-leadership with the NOAA. The membership currently includes the Puerto Rico Department of Natural and Environmental Resources (PR DNER), the Puerto Rico Environmental Quality Board (PR EQB), the U.S. Virgin Islands Department of Planning and Natural Resources (VI DPNR), the Federal Highways Administration, NOAA, the National Park Service, the United States Coast Guard, the United States Department of Agriculture's Natural Resources Conservation Service, EPA, FWS, the United States Forest Service, and the United States Geological Survey, and the U.S. Army Corps of Engineers.

The CCRP is provides a leadership forum to foster collaboration among agencies with authority/jurisdiction to respond to identified local threats. The partnership works closely and interactively with USVI and PR with a major focus on priority jurisdictional watershed projects. A very important aspect of the CCRP is to provide direct information to EPA to advance Clean Water Act protections described in the Region 2 Coral Protection Plan. Similarly, it can serve to provide input to all partner agencies to guide their plans for coral conservation and protection.

The partnership is positioned to help in responding to emerging threats. It can quickly assemble Principals and key program management to engage in executive-level discussions on emerging threats and issues.

2. EPA Region 2 Coral Protection Team

Internal communications within EPA Region 2 are maintained through the Coral Protection Team, a Region 2 team of Caribbean-focused staff from the Clean Water Division, Caribbean Environmental Protection Division and the ORD who have regular interaction with interagency partners in matters of watershed and coastal protection. Monthly conference calls are conducted raise and discuss issues and threats that need attention.

3. EPA Region 2 Coral Reef Protection Plan

In 2014, EPA Region 2 developed the Coral Reef Protection Plan in order to increase coral reef protection in the USVI and Puerto Rico. Coral reef ecosystems are being severely impacted by climate change, overfishing, pollution (including sediment runoff) and disease. In the presence of rising sea surface temperatures and ocean acidification, corals are more susceptible to adverse impacts from local stressors. To address some of these adverse impacts, the EPA Region 2 is strategically applying its regulatory and non-regulatory programs to reduce pollution (e.g. sedimentation, nutrients and pathogens) that leads to eutrophication and the degradation of coastal waters and coral reef ecosystems. Improving coral health restores their natural resilience, including the ability to better defend against climate change stresses. The 2014 Plan, revised in 2015, provides an updated strategy for EPA Region 2 in partnering and communicating with local and federal agencies involved with coral protection, and implementing a series of "direct" actions to address threats to coral reef ecosystems. It also includes activities and programs that Region 2 is targeting for future implementation. The Region reviews and revises this plan annually, updating status of the ongoing projects which include, but are not limited to, the following goals to:

- reduce point source pollution discharge in coastal waters by maximizing compliance with the schedule for attaining full-time operation of the Cruzan Rum discharge treatment system;
- reduce the amount of untreated sewage discharged to coastal waters by reduction of sewer overflows from faulty infrastructure;
- reduce the amount of sewage discharged from boats to coastal waters by setting up and implementing VI's No Discharge Zone;
- reduce nonpoint source pollution discharged to coastal waters by strengthening the effectiveness of the nonpoint source permitting program;
- reduce nonpoint source pollution discharged to coastal waters by reduction in sediment loading from construction sites;
- reduce point source storm water pollution discharges in coastal waters by reduction of unpermitted storm water discharges in areas near coral reefs by controlling PR Storm Water discharges from Municipalities, Construction Sites and Other Industrial Point Sources;

- reduce the amount of sewage discharged to coastal waters by increasing the effectiveness of new as well as existing on-site systems;
- reduce and eliminate exposure of ESA-listed species to wastewater pollutants in Vega Baja, PR by identifying areas of concern where untreated wastewater has been observed flowing into waters near *A. palmata* coral communities;
- increase public and community awareness of sanitation issues in Puerto Rico's lower Guánica Bay watershed by partnering with ORD to initiate citizen surveys of water quality and sewage infrastructure at targeted locations in watersheds adjacent to valued coral reef ecosystems at Guánica Bay; and
- maintain sustainable operations at all marinas in PR and USVI by implementation of best management practices for marina and recreational boating facilities.

4. Public Outreach

The Region 2 Public Affairs Division, with the assistance of the Coral Reef Team, is working on reaching out to local individuals and organizations to coordinate efforts to promote coral reef protection.

5. Revisions to the water quality standards to protect coral reefs

Revisions to the USVI's Water Quality Standards Regulations

During the current triennial WQS review (2015), the USVI revised its water quality standards for temperature and turbidity, making them more stringent for areas where coral reefs are located. EPA continues to work closely with the VI DPNR to encourage the adoption of additional new or revised criteria during their next triennial WQSR review processes scheduled to be completed by the end of 2018, to further improve the protection of coral reefs around the Territory.

Implementation of narrative biocriteria in the USVI

On 2010, the USVI adopted narrative biological criteria (biocriteria) into the VI's WQSR, which describe the desired condition for coral reefs, dependent on their location and associated use classification. The VI DPNR has entered into a MoA with The Nature Conservancy and has close technical ties to the University of the Virgin Islands (UVI) for technical support for coral monitoring. EPA Region 2 continues to work closely with the USVI to encourage periodic coral monitoring and, in collaboration with ORD, to develop relationships between specific water quality parameters (identified stressors) and coral condition, so that the narrative biological criteria can be implemented as a measure for ecosystem assessment and potential CWA section 303(d) listing.

Derivation of numeric biocriteria for the USVI

A long term goal for the EPA Region 2 and the VI DPNR is to derive numeric biological criteria, as well as additional numeric water quality criteria for causal parameters, such as nutrients and clean sediment. This action would allow, in the long term, for more refined assessments and decision making regarding restoration of coral reefs and attention to ESA-listed threatened and endangered species in the USVI.

Derivation of coral reef condition thresholds for the Caribbean

EPA's ORD, in collaboration with the Office of Water and Region 2, has assembled a panel of coral reef experts with expertise in coral reef taxonomic groups (e.g., stony corals, fishes, sponges, gorgonians, algae, seagrasses and macroinvertebrates), as well as community structure, organism condition, ecosystem function and ecosystem connectivity. The expert panel is developing a framework that illustrates a range of biological responses that can result from human disturbance, referenced as the coral reef biological condition gradient (BCG). The expert panel is establishing levels of condition, with a consistent well-defined narrative for each level, and a process for translating specific metric scores into levels. Levels can be aligned with designated aquatic life uses in water quality standards and can be used as targets for protection and restoration. Reef assessment data, photos and videos from federal, territorial, academic surveys and monitoring programs will be included in a US Caribbean coral reefs database that will reside on STORET database. Completion of the BCG will require a series of facilitated workshops and webinars with this group of coral reef experts. Thus far, the expert panel met in 2014 and produced a report: "*Workshop on Biological Integrity of Coral Reefs.*" The expert panel met again in October 2015. Outcomes from that work shop are being developed.

Revisions to the Puerto Rico Water Quality Standards Regulations

EPA Region 2 continues to encourage Puerto Rico to revise the PR WQSR to include new or revised narrative biological criteria and/or numeric water quality standards especially derived to protect coral reefs around PR's Islands. These actions are consistent with comments provided by NOAA during the ESA consultation process on 2010 triennial PR WQSR revisions.

VIII. Assessing effects of the U.S.V.I. Water Quality Standards on Federally-listed species

A. Background on the Derivation of criteria adopted by the U.S.V.I.

1. Dissolved Oxygen

The USVI adopted the Dissolved Oxygen (DO) criterion of no less than 5.5 mg/L for Class A and B marine waters and of no less than 5.00 mg/L for Class C marine waters. The exceptions to

the criteria are made, in both cases, when the lowered DO levels are observed due to the natural causes.

The above DO criteria have been adopted by the USVI and included in the VI's WQSR since, at least, 1985. EPA Region 2 is not in the possession of any record/documentation providing information on how this specific criterion has been derived by the VI DPNR. By personal communications with the VI DPNR staff, these criteria were based on best professional judgment after the review of available ambient water quality data and DO standards adopted by other states for similar ecosystems.

The DO criterion of not less than 5 mg/L for Class C marine waters, adopted by the USVI, is consistent with EPA's recommendation of DO criterion of not less than 5.00 mg/L, applicable to freshwaters and published in EPA's "Red Book" Report: *Quality Criteria for Water*, in 1976. EPA did not publish recommended DO criteria, which would be applicable to estuarine/marine waters at that time. EPA did not issue any recommendations for such criteria until 2000. The DO criteria adopted by the USVI in (or prior to) 1985 are consistent with these new EPA's recommendations published in the Report: *Aquatic Life Water Quality Criteria for Dissolved Oxygen (Saltwater): Cape Cod to Cape Hatteras* (EPA 2000).

The EPA's most recent DO recommendations result from a 10-year research effort. The water quality criteria represent EPA's best estimates (based on available data) of DO concentrations necessary to protect aquatic life and uses associated with aquatic life. These water quality criteria recommendations apply to coastal waters of the Virginian Province (southern Cape Cod to Cape Hatteras). However, with appropriate modification, they may be applied to other coastal regions of the United States. The recommended criteria apply to both continuous (persistent) and cyclic (diel, tidal, or episodic) hypoxia. If DO exceeds the chronic protective value for growth (4.8 mg/L), the site meets objectives for protection. If the DO is below the limit for juvenile and adult survival (2.3 mg/L), the site does not meet objectives for protection. When the DO is between these values, the site requires evaluation of duration and intensity of hypoxia to determine suitability of habitat for the larval recruitment objective. The limits identified are based entirely on laboratory findings, but are supported in part by field observations.

EPA's recommended DO criteria were derived based in the EPA's *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (Stephan et al. 1985). The approach to determine the limits of DO that will protect saltwater animals considers both continuous (i.e., persistent) and cyclic (e.g., diel) exposures to low DO and covers three areas of protection:

- juvenile and adult survival — a lower limit is calculated for continuous exposures by using the Final Acute Value (FAV) calculation procedures outlined in the *Guidelines* (Stephan et al. 1985), but with data for only juvenile or adult stages. Limits for cyclic exposures are derived from an appropriate time-to-death curve for exposures less than 24 hr. FAV is the value representing the 95th %ile genus, which for DO is 1.64 mg/L. This value is adjusted to a criterion of 2.27 mg/L by multiplying by 1.38 (the average lethal concentrations LC5 to LC50 ratio for juveniles). This value is analogous to the Criterion Maximum Concentration (CMC) in traditional Water Quality Criteria for toxicants;

- growth effect — a threshold above which long-term, continuous exposures should not cause unacceptable effects is derived from growth data (mostly from bioassays using larvae). This Final Chronic Value (FCV) is calculated in the same manner as the FAV for juvenile and adult survival. This threshold limit, as currently presented, has no time component (it can be applied to exposures of any duration). Cyclic exposures are evaluated by comparing reductions in laboratory growth from cyclic and continuous exposures. Overall, genus mean acute values (GMCVs) for effects on growth range from >1.97 mg/L for the sheepshead minnow to 4.67 mg/L for the longnose spider crab, a ratio of less than 2.4. Three of the most sensitive species were crustaceans. The range of chronic values for the four most sensitive genera is 3.97 to 4.67mg/L; and
- larval recruitment effects — a larval recruitment model was developed to project cumulative loss caused by low DO. The effects depend on the intensity and the duration of adverse exposures. The maximum acceptable reduction in seasonal recruitment was set at 5%, which is equivalent to the protective limit for juvenile and adult survival. The number of acceptable days of seasonal exposure to low DO decreases as the severity of the hypoxic condition increases. The severity of cyclic exposure is evaluated with a time-to-death model (as in the protective limit for juveniles and adults).

Recommended by EPA in 2000, DO criteria apply to both continuous and cyclic low DO conditions. If the DO conditions are always above the chronic criterion for growth (4.8 mg/L), the aquatic life at that location should not be harmed. If the DO conditions at a site are below the juvenile/adult survival criterion (2.3 mg/L), there is not enough DO to protect aquatic life.

In summary, criteria for DO adopted by the USVI are believed to be consistent with the most recent EPA recommendations and thus, are considered to be protective of aquatic life uses in the USVI.

2. pH

The USVI has adopted the pH criterion of no less than 7.0 or greater than 8.3 for Class A and B marine waters and not less than 6.7 or greater than 8.5 for Class C marine waters. In both cases, the normal range of pH must not be extended at any location by more than +/- 0.1 pH unit.

The pH criteria have been adopted based on the VI DPNR review of standards adopted by other entities for similar ecosystems, using all available ambient water quality data, and the best professional judgment of the technical advisory group comprised of professionals.

The pH criteria adopted by the USVI are within the recommended range and consistent with the EPA recommendations published in 1976 and described in the EPA's "Red Book": *Quality Criteria for Water*. EPA recommended pH criterion of no less than 6.5 or greater than 8.5 for marine waters. For open ocean waters, where the depth is substantially greater than the euphotic zone, the pH should not be changed more than 0.2 units outside of the naturally occurring variation or in any case outside the range of 6.5 to 8.5. For shallow, highly productive coastal and estuarine areas, where naturally occurring pH variations approach the lethal limits for some species, changes in pH should be avoided but in any case not exceed the limits established for fresh waters ranging from 6.5 to 9.0.

The chemistry of the marine waters is unique due to presence of high concentrations of salts, alkalinity based on the carbonate systems and weak acid salts (borate). In general, the naturally occurring variability of pH in the sea water is observed to a smaller degree than in fresh waters. Studies indicate that plankton and benthic invertebrates are more sensitive than fish to changes in pH. In addition, mature forms and larvae of oysters were shown to be adversely affected at the extremes of the pH range of 6.5 to 9.00. However, in the shallow, biologically active waters in tropical and sub-tropical areas, large diurnal pH changes occur naturally because of photosynthesis. The pH values may range from 9.5 in the daytime to 7.3 in the early morning before dawn.

In summary, criteria for pH adopted by the USVI, are within the range recommended by the EPA and thus, are considered to be protected of the aquatic life use in the USVI.

3. Temperature

For areas of Class B and C waters, where coral reefs are not present, the USVI adopted the temperature criterion of not to exceed 32 degrees Celsius (C) at any time. In both cases, the temperature resulting from the waste discharge must not be greater than 1 degree C above natural. In addition, the USVI thermal policies apply to both classes of water.

For all of the Class A waters and the areas of Class B and C waters where coral reef ecosystems are located, the USVI adopted a temperature criterion which shall not exceed the range 25 to 29 degrees C at any time. The temperature resulting from waste discharge must not be greater than 1 degree C above natural. In addition, the USVI thermal policies apply to Class A waters.

The temperature criterion of not to exceed 32 degrees C, at any time, was adopted based on EPA's recommended 304(a) aquatic life criterion published in 1976 and described in EPA's "Red Book" Report: *Quality Criteria for Water* (EPA 1976). Life associated with the aquatic environment in any location has its species composition and activity regulated by water temperature. The temperature of the water regulates their metabolism and ability to survive and reproduce effectively. Through the natural changes in climatic conditions, the temperatures of water bodies fluctuate daily, as well as seasonally. These changes do not eliminate indigenous aquatic populations, but affect the existing community structure and the geographic distribution of species. Such temperature changes are necessary to induce the reproductive cycles of aquatic organisms (from gametogenesis to spawning) and to regulate other life factors. Fish migration is also often linked to natural environmental temperature cycles.

Juvenile and adult fish usually thermoregulate behaviorally by moving to water having temperatures closest to their thermal preference. This provides a thermal environment, which approximate the optimal temperature for many physiological functions, including growth. Avoidance will occur as warmer temperature exceeds the preferred conditions by 1 to 3 degrees C.

Thermal discharges should not alter diurnal and tidal temperature variations normally experienced by marine communities. Laboratory studies show thermal tolerance to be enhanced when animals are maintained under a diurnally fluctuating temperature regime rather than at a constant temperature. A daily cyclic regime can be protective additionally as it reduces duration of exposure to extreme temperatures.

Summer thermal maxima should be established to protect the various marine communities within each biogeographic region. During the summer, naturally elevated temperatures may be of sufficient magnitude to cause death or emigration. This more commonly occurs in tropical and warm temperate zone waters, but has been reported for enclosed bays and shallow waters in other regions as well. Summer heat stress also can contribute to increased incidence of disease or parasitism; reduce or block sexual maturation; inhibit or block embryonic cleavage of larval development; reduce feeding and growth of juveniles and adults; result in increased predation and reduce productivity of macroalgae and seagrasses.

The USVI adopted EPA's 304(a) recommended criterion for temperature of not to exceed 32 degrees C at any time for class B and C waters,outside of areas where coral reefs are located.

To provide more adequate protection for the sensitive coral reefs species, for the areas where such ecosystems are located, the USVI adopted more stringent temperature criteria of not to exceed the range of 25 to 29 degrees C, at any time. This criterion is based on NOAA's recommendations. As described by NOAA, reef-building corals are restricted in their geographic distribution by their physiology (NMFS 2001). Reef building corals cannot tolerate water temperatures below 18 degrees C, while some species can tolerate temperatures as high as 40 degrees C, for short periods. In general, however, many species grow optimally in water temperatures between 23 to 29 degrees C (71 FR 26852). This recommendation was also referenced by NOAA in the letter to EPA, dated December 2, 2010, related to the ESA consultation initiated for the USVI’s WQSR.

4. Turbidity

For all of the Class A waters and the areas of Class B and C waters where coral reef ecosystems are located, the USVI adopted a maximum permissible nephelometric turbidity unit reading of one (1). In order to provide more adequate protection for the sensitive coral reef species, this criterion was added to the VI WQSR and adopted by the USVI in 2015 as a result of the NOAA-NMFS recommendations provided to the EPA Region 2 and VI DPNR during the ESA consultation on VIWQSR in 2010.

For areas of Class B and C where coral reefs are not present, the USVI adopted a maximum permissible nephelometric turbidity unit (NTU) reading of three (3). This turbidity criterion has been adopted by the USVI and included in the VI's WQSR since, at least, 1985. EPA Region 2 is not in possession of any record/documentation providing information on how this specific criterion has been derived by the VI DPNR. Based on personal communications with the VI DPNR staff, it had been established that this criterion was based best professional judgment after review of available ambient water quality data and turbidity standards adopted, at that time, by

other states for similar ecosystems. Although the basis for the derivation of this criterion of 3 NTUs is not known, it is important to point out that this criterion is more stringent than the EPA's recommended turbidity standard of not to exceed 10 NTUs, which was published in EPA's "Green Book" (EPA 1968).

In summary, criteria for turbidity adopted by the USVI are consistent with (or more stringent than) EPA's and NOAA's recommendations, thus are considered to be protective of the aquatic life uses in the USVI.

5. Clarity

For all marine waters (class A, B and C waters) the USVI adopted a clarity criterion, where a secchi disk shall be visible at a minimum depth of one (1) meter. For waters where the depth does not exceed one (1) meter, the bottom must be visible.

This criterion for clarity has been adopted by the USVI and included in the VI WQSR since, at least, 1985. EPA Region 2 is not in the possession of any record/documentation providing information on how this specific criterion has been derived by the VI DPNR. Based on the personal communications with the VI DPNR staff, it had been established that this criterion was based on best professional judgment after the review of available at that time ambient water quality data and clarity standards adopted by other states for similar ecosystems.

6. Phosphorus

For all marine waters (class A, B and C waters) the USVI adopted a criterion for total phosphorus (TP) of not to exceed 50 μg/L.

This TP criterion has been adopted by the USVI and included in the VI WQSR since, at least, 1985. EPA Region 2 is not in possession of any record/documentation providing information on how this specific criterion has been derived by the VI DPNR for protection of aquatic life uses applicable to marine waters. Based on personal communications with the VI DPNR staff, it had been established that this criterion, most likely, was based on the EPA's recommended 304(a) aquatic life criterion published in 1976 and described in EPA's "Red Book" Report: *Quality Criteria for Water* (EPA 1976). This TP criterion of not to exceed 50 μg/L was recommended by EPA for any stream or other flowing waters to prevent the development of biological nuisances and to control accelerating cultural euthrophication. There were no recommendations made by EPA at that time for TP criterion applicable to estuarine or marine ecosystems.

Based on the TP criteria adopted presently by other states to protect aquatic life uses in similar marine ecosystems (e.g. TP criteria for estuaries in Florida, adopted in 2014), EPA considers the USVI's TP criterion of 50 μg/L to be protective. EPA is presently working with the VI DPNR to develop Total Nitrogen (TN) criterion and to reevaluate the existing TP criterion to ensure the

appropriate protection of sensitive coral reefs ecosystems. The USVI plans to adopt more stringent TP criterion (if justified) and a new TN criterion in 2018.

B. Background on the Derivation of EPA 304a criteria adopted by the U.S.V.I.

1. General EPA procedures

Section 304(a)(l) of the CWA requires that the Administrator of the EPA publish water quality criteria that accurately reflect the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare that might be expected from the presence of pollutants in any body of water, including ground water. As EPA publishes the recommended criteria for the individual pollutants, a final ambient water quality criteria (AWQC) document is being published based upon consideration of all available information relating to effects of an individual pollutant on aquatic organisms.

During development of CWA section 304(a) criteria, EPA assembles available test data and considers all the relevant data that meet acceptable data quality standard together for all genera. In most cases, data on freshwater and estuarine/marine species are grouped separately to develop separate freshwater and estuarine/marine criteria. Thus, where data allow, four criteria are developed (acute value for freshwater, acute value for estuarine/marine waters, chronic value for freshwater, and chronic value for estuarine/marine waters). If plants are more sensitive than vertebrates and invertebrates, plant criteria are developed and recommended for adoption.

Once the section 304(a) water quality criteria are finalized, states and authorized tribes may adopt the criteria into their WQSR to protect designated uses of water bodies. States and tribes may also modify the criteria to reflect site-specific conditions or use other scientifically defensible methods to develop their own standards. Subsequently, all of the water quality standards being adopted by state are approved by EPA.

EPA derives ambient water quality criteria for the protection of aquatic life that are protective of the designated uses established for waters of the US. In this process, EPA is using the peer-reviewed procedures defined in the Agency's *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (Stephan et al. 1985).

The chronic and acute criteria recommended by the EPA build from the following principles set forth in the 1985 Guidelines:

- Acute toxicity test data must be available for species from a minimum of eight diverse taxonomic groups. The diversity of tested species is intended to assure protection of various components of an aquatic ecosystem. The Final Acute Value (FAV), which represents Lethal Concentration LC50 or Effective Concentration EC50, is derived by extrapolation or interpolation to a hypothetical genus more sensitive than 95 % of all

tested genera. The FAV is divided by two in order to obtain an acute criterion protective of nearly all individuals in such a genus;

- Chronic toxicity test data (longer-term survival, growth, or reproduction) must be available for at least three taxa. If chronic values are available for specified number and array of species, a Final Chronic Value (FCV) can be calculated directly, using the same procedure as used for the FAV derivation. If not, an acute-chronic ratio (ACR) is derived and then used with the FAV to obtain the FCV;

- The Final Plant Value (FPV) is obtained by selecting the lowest plant toxicity value based on measured concentrations;

- The Final Residue Value (FRV) is intended to protect wildlife which consume aquatic organisms and the marketability of aquatic organisms. Two kinds of data are necessary to calculate FRV; bioconcentraction factor (BCF) and a maximum permissible tissue concentration, which can be an FDA action level or can be the result of the chronic wildlife feeding study. For lipid-soluble pollutants, the BCF is normalized for % lipids and then the FRV is calculated by dividing the maximum permissible tissue concentration by the normalized BCF and by an appropriate % lipid value;

- If sufficient data are available to demonstrate that one or more of the final values should be related to a water quality characteristic (e.g. salinity, hardness, or suspended solids), the final value(s) are expressed as a function of that characteristic;

- After the four final values (FAC, FCV, FPV, and FRV), have been obtained, the criterion is established with the FAV becoming the maximum value and the lowest of the other three values becoming the 24-hour average value;

- All of the data used to calculate the four final values and any additional pertinent information are then reviewed to determine if the criterion is reasonable. If sound scientific evidence indicates that the criterion should be raisen or lowered, appropriate changes are made, as necessary;

- When evaluating tune-variable ambient concentrations generally, one-hour average concentration are considered to be appropriate for comparison with the acute criterion, and four-day averages with the chronic criterion; and

- The allowable frequency for exceeding a criterion is set at once every three years, on the average.

Each assessment endpoint requires one or more “measures of ecological effect” which are defined as changes in the attributes of an assessment endpoint itself or changes in a surrogate entity or attribute in response to chemical exposure. Ecological effect data are used as measures of direct and indirect effects to biological receptors. The measures of effect selected represent the growth, reproduction, and survival of the organisms.

- The acute measures of effect used for organisms in this document are the LC50 (the concentration of a chemical that is estimated to kill 50% of the test organisms), EC50 (the concentration of a chemical that is estimated to produce a specific effect in 50 % of the test organisms) and the IC50 (the inhibitory concentration of a chemical that is estimated to inhibit some biological process (i.e. growth, etc.) by 50 % compared to a control organism).

- Endpoints for chronic measures of exposure are the NOEC, LOEC, and MATC. The NOEC (i.e., "No-Observed-Effect-Concentration") is the highest test concentration at which none of the observed effects were statistically different from the control. The LOEC (i.e., "Lowest-Observed-Effect-Concentration") is the lowest test concentration at which observed effects were statistically different from the control. The Maximum Acceptable Toxicant Concentration (MATC) is the calculated geometric mean of the NOEC and LOEC.

The CWA criteria are based on a species sensitivity distribution (SSD) comprised of genus mean acute values (GMAVs), calculated from species mean acute values (SMAVs) for acceptable available data. SMAVs are calculated using the geometric mean for all acceptable toxicity tests within a given species (e.g. all tests for *Daphnia magna*). If only one test is available, the SMAV is that test value by default. GMAVs are then calculated using the geometric means of all SMAVs within a given genus (e.g. all SMAVs for genus Daphnia - *Daphnia pulex, Daphnia magna*). Once again, if only one SMAV is available for a genus, then the GMAV is represented by that value. GMAVs are then rank-ordered by sensitivity from most sensitive to least sensitive. The FAV is determined by regression analysis based on the four most sensitive genera (reflected as GMAVs) in the data set to interpolate or extrapolate (as appropriate) to the 5th %ile of the distribution represented by the tested genera. The acute criterion is the FAV divided by two, which is intended to provide an acute criterion protective of nearly all individuals in such a genus.

2. Procedures to derive Acute Value

To derive an acute criterion for saltwater aquatic organisms and their uses, the following should be available:

1. Results of acceptable acute tests with at least one species of saltwater animal, in at least eight different families, such that all of the following are included:

   - two families in the phylum *Chordata*;
   - a family in a phylum other than *Arthropoda* or *Chordata*;
   - either the *Mysidae* or *Penaeidae* family;
   - three other families not in the phylum *Chordata* (may include *Mysidae* or *Penaeidae*, whichever was not used above); and
   - one representative of any other family.

2. ACRs with species of aquatic animals, in at least three different families, provided that of the three species at least one is a fish, at least one is an invertebrate, and at least one is an acutely sensitive saltwater species (the other two may be freshwater species);

3. Results of at least one acceptable test with a saltwater alga or vascular plant. If plants are among the aquatic organisms most sensitive to the material, results of a test with a plant in another phylum (division) should also be available; and

4. At least one acceptable BCF determined with an appropriate saltwater species, if a maximum permissible tissue concentration is available.

For each species for which at least one acute value is available, the SMAV should be calculated as the geometric mean of the results of all flow-through tests in which the concentrations of test material were measured.

Appropriate measures of the acute (short-term) toxicity of the material to a variety of species of aquatic animals are used to calculate the FAV. The FAV is an estimate of the concentration of the material corresponding to a cumulative probability of 0.05 in the acute toxicity values for the genera with which acceptable acute tests have been conducted on the material. However, in some cases, if the SMAV of a commercially or recreationally important species is lower than the calculated FAV, then that SMAV replaces the calculated FAV in order to provide protection for that important species. The Criterion Maximum Concentration (CMC) is equal to one-half the FAV.

As it is specified in the "Red Book" (EPA 440/9-76-023), in cases where only 96-hour bioassay data are available, EPA recommends the application of the substantial safety factor to protect all life stages of the test organism in water of varying quality, as well as to protect associated organisms within the aquatic environment that have not been tested and that may be more sensitive to the test constituent. In such cases, application factors are to be used to provide the degree of protection required. Safe levels for certain chlorinated hydrocarbons and certain heavy metals were estimated by EPA by applying 0.01 application factor to the 96-hour LC50 value for sensitive aquatic organisms.

3. Procedures to derive Chronic Value

The chronic criterion may be determined by one of two methods. If all eight minimum data requirements are met with acceptable chronic test data, then the chronic criterion is derived using the same method used for the acute criterion. In cases where less chronic data are available (i.e., must have at least three chronic tests from taxa that also have appropriate acute toxicity data) the chronic criterion can be derived by determining an appropriate ACR.

Chronic values should be based on endpoints and lengths of exposure appropriate to the species. A chronic value may be obtained by calculating the geometric mean of the lower and upper chronic limits from a chronic test or by analyzing chronic data using regression analysis. A lower chronic limit is the highest tested concentration in an acceptable chronic test, which did

not cause an unacceptable amount of adverse effect on any of the specified biological measurements, and below which no tested concentration caused an unacceptable effect. An upper chronic limit is the lowest tested concentration in an acceptable chronic test, which did cause an unacceptable amount of adverse effect on one or more of the specified biological measurements, and above which all tested concentrations also caused such an effect.

If chronic values are available for species in eight families, a SMCV should be calculated for each species for which at least one chronic value is available by calculating the geometric mean of all chronic values available for the species, and appropriate GMCV should be calculated. The FCV should then be obtained.

For each chronic value for which at least one corresponding appropriate acute value is available, calculate an acute-chronic ratio, using for the numerator the geometric mean of the results of all acceptable flow-through (except static is acceptable for daphnids) acute tests in the same dilution water and in which the concentrations were measured. For each species, calculate the species mean acute-chronic ratio as the geometric mean of all ACRs available for that species. Calculate the Final ACR as the geometric mean of the available species mean ACRs.

The Criterion Continuous Concentration (CCC) is equal to the lowest of the FCV, the Final Plant Value, and the Final Residue Value, unless other data show that a lower value should be used. If toxicity is related to a water quality characteristic, the CCC is obtained from the Final Chronic Equation, the Final Plant Value, and the Final Residue Value by selecting the one, or the combination, that results in the lowest concentrations in the usual range of the water quality characteristic, unless other data show that a lower value should be used.

In cases where sufficient data is not available, the chronic criterion can also be derived by determining an appropriate ACR. The ACR is a way of relating the acute and chronic toxicities of a material to aquatic organisms. ACRs can be used to derive chronic criteria with data for species of aquatic animals provided that at least three of the minimum data requirements are met and that: (1) at least one is a fish, (2) at least one is an invertebrate and (3) at least one is an acutely sensitive estuarine/marine species; the other two species data may be freshwater or estuarine/marine as appropriate to the derivation. ACRs are calculated by dividing the acute toxicity test values by a "paired" (same lab, same dilution water) chronic test value. Comparisons of ACRs across taxa may elucidate differences and similarities in taxa response. If variability is greater than ten-fold among calculated ACRs, and no explainable trend exists, then a chronic criterion should not be derived. The Final ACR (FACR) is then derived by calculating the geometric mean of all acceptable ACRs. The FCV is then estimated by dividing the FAV by the FACR. This serves as the basis for the chronic criterion. Finally, the acute or chronic criterion may be lowered to protect recreationally or commercially important species.

In the WQSR, section 186-5(b)(1)(a), the USVI adopted the following frequency and duration provisions to clarify how the adopted aquatic life criteria are expressed:

1. Acute aquatic life protection criteria are expressed as one-hour average not to be exceeded more than once over a three year period.

2. Chronic aquatic life protection criteria are expressed as four-day average not to be exceeded more than once over a three year period.

5. For ammonia, the highest four-day average within the 30-day period should not exceed 2.5 times the CCC.

The above listed frequency and duration components are applicable to all of the aquatic life criteria, unless otherwise stated. The recommended frequency of exceedance, once in 3 years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from an event in which exposure to pollutant exceeds the criterion.

4. Derivation of 304a Recommended Criteria for Toxic Pollutants

a) Organic Compounds

(1) Pesticides/Herbicides

(a) Aldrin and Dieldrin

Both, aldrin and dieldrin are organochlorine insecticide and are no longer produced or used. Between1950s and 1970, aldrin and dieldrin were used extensively for crops such as corn and cotton. The USDA canceled all uses of both insectcidesin 1970.

The USVI adopted an aldrin criterion of 1.3 µg/L (acute value) for protection of saltwater aquatic life in all marine waters (class A, B and C waters). The USVI adopted dieldrin criteria of 0.71 µg/L (acute value) and 0.0019 µg/L chronic value for protection of saltwater aquatic life in all marine waters (class A, B and C waters). All three criteria were adopted by the USVI based on EPA's recommended 304(a) aquatic life criterion, published in EPA's Report: *Ambient Water Quality Criteria for Aldrin/Dieldrin* (EPA 1980h). In the past, aldrin and dieldrin have been two of the most widely used domestic pesticides. These pesticides are often considered together because in the environment, aldrin is rapidly converted to a more stable (and at least equally toxic) form – dieldrin, accomplished through the addition of an epoxide group to the aldrin molecule. Since aldrin is rapidly converted to dieldrin, no chronic criterion has been developed for aldrin.

Saltwater invertebrate species were acutely sensitive to both aldrin and dieldrin, but there were greater differences in reported LC50 values for these species than for saltwater fish species. Saltwater invertebrate acute values ranged from 0.37 to 33µg/L for aldrin and ranged from 0.28 to 50µg/L for dieldrin. The most sensitive species to aldrin in a 96-hour test was Korean shrimp with LC50 values of 0.74 and 3.0 µg/L. The commercially important pink shrimp was the most sensitive species to dieldrin in a 96-hour test with an LC50 value of 0.7µg/L. Other invertebrate species were less sensitive to dieldrin, and their acute LC50 values ranged from 3.7 to 50 µg/L.

All species of saltwater fishes tested were sensitive to acute exposures to aldrin or dieldrin. In aldrin exposures, the 96-hour LC50 values for 11 fish species ranged from 2.03 μg/L (for dwarf perch) to 100 μg/L (for striped mullet). The acute LC50 values for 13 fish species exposed to dieldrin ranged from 0.9 μg/L (for American eel) to 34.0 μg/L (for northern puffer). Generally, the LC50 values for aldrin were slightly higher than those for dieldrin in tests where the same species were tested.

Based on SMAVs, the saltwater FAV for dieldrin was 0.71 μg/L, as calculated according to the procedure described in the EPA's *Guidelines*, and that for aldrin was1.3 μg/L.

No chronic study on any saltwater fish species had been reported. The only chronic data found for saltwater species was a 28-day life cycle study on the mysid shrimp with dieldrin. In that study, the chronic limits were 0.49 and 1.1 μg/L based on cumulative mortality. Effects on reproduction were not observed in any of the test concentrations. The geometric mean of these two values, 0.73 μg/L, became the chronic value for mysid shrimp. Dividing this value into the acute value for this species of 4.5 μg/L gave an ACR of 6.2. The FACR for dieldrin of 8.5 was the geometric mean of the three ACRs. The saltwater FAV for dieldrin of 0.71 μg/L divided by the FACR of 8.5 resulted in the saltwater FCV for dieldrin of 0.084 μg/L.

Dieldrin BCF for saltwater species ranged from 400 to 8,000. The Saltwater FRV of 0.0019 μg/L was calculated using the FDA action level of 0.3 mg/kg of fish oil, a % lipid value of 100 for fish oil, and the geometric mean of normalized BCFs. The USVI adopted the above described aldrin and dieldrin criteria into the VIWQSR.

(b) Carbaryl

Carbaryl (1-naphthyl methylcarbamate) is a chemical in the carbamate family used primarly as an insecticide. It remains the third-most-used insecticide in the United States for home gardens, commercial agriculture, and forestry and rangeland protection.The USVI adopted a carbaryl criterion of 1.6 μg/L (acute value) for protection of saltwater aquatic life in all marine waters (class A, B and C waters).

The criterion was adopted by the USVI based on EPA's recommended 304(a) aquatic life criterion published in 2012 and described in EPA's Report: *Aquatic Life Ambient water Quality Criteria for Carbaryl* (EPA 2012). The criterion for carbaryl was derived using the *Guidelines* (Stephan et al. 1985).

The carbaryl aquatic life criteria had been developed by the EPA using peer reviewed methods and data that were acceptable for the derivation of a freshwater and estuarine/marine criteria. Data evaluated for criteria derivation include data submitted in support of the registration of this pesticide and reviewed by U.S. EPA's Office of Pesticide Programs as well as studies reported in the open literature and identified in a literature search using the ECOTOXicology database (ECOTOX) as meeting data quality standards. ECOTOX is a source of high quality toxicity data for aquatic life, terrestrial plants, and wildlife. The database was created and is maintained by

the EPA's ORD and the National Health and Environmental Effects Research Laboratory's Mid-Continent Ecology Division.

The assessment endpoints for carbaryl criteria to protect aquatic life were based on growth, reproduction, and survival of the assessed taxa. The measures of effect were provided by the acute and chronic toxicity data. These toxicity endpoints, expressed as genus means, were used in the species sensitivity distribution (SSD) of the aquatic community to derive the aquatic life criteria.

Acute toxicity data for carbaryl were available for 12 estuarine/marine species representing 11 genera. These data represented a dataset supporting the development of an estuarine/marine acute criterion. SMAVs for carbaryl ranged from 7.188 to 17,000 µg/L. The most sensitive genus was the mysid (*Americamysis*), with a GMAV of 7.188 µg/L, followed by the Dungeness crab (*Metacarcinus*) with a GMAV of 10 µg/L. The two most tolerant genera were the bent-nosed clam (*Macoma*) and the threespine stickleback (*Gasterosteus*), with SMAVs of 17,000 and 3,990 µg/L, respectively.

The *Guidelines* indicate that eight minimum data requirements are needed to calculate an estuarine/marine FAV; data were available for 11 genera and met the family requirement outlined above. The estuarine/marine FAV was 3.15 µg/L. The estuarine/marine CMC (1.58 µg/L) was protective of all estuarine/marine organisms acutely exposed to carbaryl.
The resulting criterion for carbaryl to protect saltwater aquatic life, as derived using the *Guidelines,* is the 1-hour average concentration not exceeding 1.6 µg/L more than once every three years on average (acute value). There was no sufficient data available to derive the chronic value for carbaryl. The USVI has adopted this criterion into the WQSR.

(c) Chlordane,

Chlordane, or chlordan, is an organochlorine compound used as a pesticide. The USVI adopted a chlordane criteria of 0.09 µg/L (acute value) and 0.004 µg/L (chronic value) for protection of aquatic life in all marine waters (class A, B and C waters). These criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 1980 and described in EPA's Report: *Ambient Water Quality Criteria for Chlordane* (EPA 1980b).

Acute chlordane values for saltwater invertebrate species ranged from 0.4 to 480 µg/L, with the pink shrimp being the most sensitive species. Blue crab, in a 48-hour test, was over 1,000 times more tolerant than pink shrimp in a 96-hour test. Adult Dungeness crabs were also tolerant of acute chlordane exposure, with an LC50 value of 220 µg/L, but Dungeness crab zoeae had a much lower LC50 of 1.3 µg/L. Five species of saltwater fishes had been tested for the acute effects of chlordane. In flow-through exposures, the 96-hour LC50 values for three species ranged from 6.4 to 24.5 µg/L. Two LC50 values for three spine stickleback from static tests with unmeasured concentrations were 90 and 160 µg/L. The LC50 values for fish species differed by a factor of more than 25.

The minimum data base requirements for deriving a saltwater FAV had not been met (96-hours LC50 values were available for four, instead of the required five invertebrate families); however, data were available for eight species (four invertebrate and four fish species). Moreover, it is unlikely that the saltwater FCV would be significantly influenced by one more acute value for an invertebrate species. Accordingly, a saltwater FAV for chlordane of 0.09 μg/L was derived from the SMAV, using the procedure described in the *Guidelines*. In extended exposures of Dungeness crab zoeae and adults to chlordane, 0.15 μg/L was lethal to 50 % after 37 days of exposure. Survival was unaffected in chlordane concentrations of 0.015 μg/L. Most adult crabs died after a 90-day continuous exposure to 10 μg/L.

Chlordane was chronically toxic to the saltwater sheepshead minnow in a full life-cycle-exposure at concentrations 0.8 μg/L. Survival of juveniles was reduced at 18 μg/L, and their survival through adulthood was reduced at 2.8 μg/L. Reproduction of exposed adults was not impaired, but hatching of embryos was decreased at 0.8 μg/L, and juvenile survival decreased at 1.7 μg/L. No significant effects were observed on survival, growth, or reproduction at a chlordane concentration of 0.5 μg/L. A life-cycle chronic test on the sheepshead minnow provided a chronic value for this species of 0.63 μg/L. No chronic data were available for chlordane and any saltwater invertebrate species. The FACR for chlordane of 14 was the geometric mean of the three ACRs. The Saltwater FAV of 0.09 μg/L divided by the FACR of 14 resulted in the saltwater FCV for chlordane of 0.0064 μg/L.

The data base for acute toxicity of chlordane to saltwater species was missing one invertebrate family to fulfill the minimum data base requirements according to the *Guidelines*. However, because acute data was available for eight species and because it was unlikely that the saltwater FCV (calculated to be 0.0064 μg/L) would be significantly influenced by one additional acute value for an invertebrate species, a saltwater FAV was derived for chlordane and was calculated to be 0.09 μg/L.

Whole-body BCF values for the saltwater fish species, sheepshead minnow, ranged from 6,600 to 16,000. The BCF in juvenile fish ranged from 8,500 to 12,300 after 28 days of exposure to technical chlordane. Adult fish exposed to technical chlordane for 189 days had BCF values ranging from 13,000 to 22,000. Dividing a BCF value by the % lipid value for the same species provides a BCF value adjusted to 1 % lipid content; this resultant BCF value was referred to as the normalized BCF. % lipid values were available for fathead minnows and adult sheepshead minnows. Dividing the % lipid value of 7.6 for fathead minnows into the

BCF of 37,800 gave a normalized BCF of 4,974. Dividing the % lipid value at 3.6 for sheepshead minnows into the BCF of 16,000 gave a normalized BCF of 4,444. The geometric mean of these normalized BCF values was 4,702.

To protect the marketability of edible fish, the concentration of chlordane in edible tissue cannot exceed the action level of 0.3 mg/kg established by the U.S. Food and Drug Administration (FDA) for chlordane. The Saltwater FRV was 0.0040 μg/L, obtained by dividing the FDA action level (0.3 mg/kg) by the geometric mean of normalized BCF values (4,702) and by a % lipid value of 16 for saltwater species.

In summary, EPA's 304(a) aquatic life criterion for chlordane, the chronic criterion to protect saltwater aquatic life was calculated to be 0.004 μg/L (based on the FRV), as a 24-hour average and the concentration should not exceed 0.09 μg/L at any time. The USVI had adopted the above criteria for chlordane into the WQSR.

(d) Chlorpyrifos

Chlorpyrifos ia a organophosphate insecticide. Although most use in homes has been banned since 2001 in the U.S., in agriculture, it remains one of the most widely used organophosphate insecticides.

The USVI adopted a chlorpyrifos criteria of 0.011 μg/L (acute value) and 0.0056 μg/L chronic value for protection of saltwater aquatic life in all marine waters (class A, B and C waters). Both criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 1986 and described in EPA's Report: *Ambient Water Quality Criteria for Chlorpyrifos* (EPA 1986c). Both criteria for chlorpyrifos were derived using the *Guidelines* (Stephan et al. 1985).

Tests of the acute toxicity of chlorpyrifos to saltwater animals had been conducted with 15 species of saltwater animals in 12 genera (species of invertebrates and ten species of fish). The range of acute values was reported from 0.01 μg/L for adult Korean shrimp to 1,991 μg/L for larvae of the eastern oyster. Four species of saltwater arthropods had been tested and they were all more sensitive than the most sensitive fish species. The range of acute toxicity values for fish was narrower than for invertebrates, with LC50s extending from 0.4 μg/L for 14-day-old larvae of the tidewater silverside to 520 μg/L for juveniles of the gulf toadfish.

A series of 96-hr acute tests were conducted under both static and flow-through conditions with four different ages of larvae of three estuarine fishes. LC50s ranged from 0.4 to 5.5 μg/L for all tests. In static tests, 14-day-old larvae were more sensitive than newly hatched or 28-day larvae of all species. In flow through tests, relative sensitivities of the ages were similar to those in static tests' for tidewater silverside, decreased with age for Atlantic silverside, and differed little for California grunion.

Of the twelve genera for which saltwater GMAV are available, the most sensitive genus, *Mysidopsis*, was about 57,000 times more sensitive than the most resistant genus, *Crassostrea*. Acute values were available for more than one species in each of two genera, and the range of SMAVs within each genus was less than a factor of 5.7. The saltwater FAV was calculated to be 0.02284 μg/L, which is lower than the lowest GMAV. Following up the procedures identified in the EPA's Guidance (Stephan et al. 1985), CMC of 0.011 μg/L was calculated by dividing the FAV by 2.

Data on the chronic toxicity of chlorpyrifos to saltwater animals were available for the mysid, *Mysidopsis bahia*, and six fishes. In the 28-day life-cycle test with the mysid, survival and reproduction were reduced at 42 μg/L, and growth was significantly reduced at a nominal concentration of 0.004 μg/L. Of the six saltwater fishes exposed to chlorpyrifos in early life-stage toxicity tests, the California grunion was the most sensitive. Decreased weight was the

most sensitive endpoint for this species, the sheepshead minnow, and the gulf toadfish. Decreased survival was the most sensitive endpoint with the three species of Menidia, although growth was also affected with two of these species.

The Species Mean ACRs for the seven saltwater species ranged from 1.374 to 228.5. However, the ratios for the five sensitive species only ranged from 1.374 to 12.50. Thus, the Final ACR for chlorpyrifos was calculated as the geometric mean of ratios derived for these five species. Division of the saltwater FAV by the Final ACR of 4.064 resulted in FCV of 0.00562 μg/L. The saltwater value is a factor of two higher than the chronic value for the most acutely sensitive saltwater species, *Mysidopsis bahia.*

The procedures described in the *Guidelines,* indicate that saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of chlorpyrifos does not exceed 0.0056 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.011 μg/L more than once every three years on the average. The recommended exceedance frequency of 3 years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to chlorpyrifos exceeds the criterion. The USVI has adopted the above criteria for chlorpyrifos into the WQSR.

(e) Demeton

Demeton is a phosphorothioate insecticide. The USVI adopted a demeton criterion of 0.1 μg/L chronic value for protection of saltwater aquatic life in all marine waters (class A, B and C waters). This criterion was adopted based on EPA's recommended 304(a) aquatic life criterion published in 1976 and described in EPA's "Red Book" Report: *Quality Criteria for Water* (EPA 1976).

There were few data on the toxicity of demeton to marine organisms. A 48-hour EC50 of 63 μg/L was reported for the pink shrimp, *Peneaus duorarum*, and a 24-hour LC50 of 550 μg/L was reported for the spot, *Leiostomus xanthurus*. Criterion was derived based partly on the fact that all organophosphates (including demeton) inhibit the production of the acetylcholinesterase (AChE) enzyme. Demeton is unique, however, in that the persistence of its AChE inhibiting ability was greater than that of 10 other common organophosphates, even though its acute toxicity was apparently less. The effective "half-life" of AChE inhibition for demeton was greater than one year, which could have been additive with repeated exposures and could have been compounded by any of the organophosphates. This was a reason why recommendation was made that criterion for demeton be based primarily on its enzyme inhibiting potential. A criterion of 0.1 μg/L demeton for freshwater and marine aquatic life was recommended since it will not be expected to significantly inhibit AChE over a prolonged period of time. In addition, the criteria recommendation was in close agreement with the criteria for the other organophosphates.

In summary, the EPA's 304(a) recommendation is that, for demeton, the criterion to protect saltwater aquatic life is 0.1 μg/L (chronic value). The USVI has adopted this criterion for demeton into the WQSR.

(f) Diazinon

Diazinon is organophosphate insecticide formerly used to control cockroaches, silverfish, ants, and fleas in residential buildings. Although the residential use of diazinon was outlawed in the U.S. in 2004, it is still approved for agricultural uses.

The USVI adopted a diazinon criterion of 0.82 μg/L acute and chronic value for protection of saltwater aquatic life in all marine waters (class A, B and C waters). This criterion was adopted based on EPA's recommended 304(a) aquatic life criterion published in 2005 and described in EPA's Report: *Ambient Water Quality Criteria for Diazinon* (EPA 2005b). The criterion for diazinon was derived using the *Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses* (Stephan et al. 1985).

The acute toxicity of diazinon to saltwater animals had been determined for 7 invertebrate species and 2 fish species. SMAVs ranged from 2.57 μg/L for the copepod to > 9,600 μg/L for embryos of the sea urchin. Acute values for the mysid, determined from a renewal, unmeasured test (8.5 μg/L) were approximately two-fold higher than those determined from a flow-through measured test (4.82 μg/L). Acute toxicity test data available for other saltwater invertebrates included an annelid worm, an amphipod and two species of shrimp (grass shrimp and pink shrimp). The saltwater fish, inland silverside was relatively insensitive to diazinon with a LC50 of 1,170 μg/L. The remaining fish species, the sheepshead minnow had an LC50 value of 1,400 μg/L, and is the only saltwater fish with a corresponding chronic value. Acute values for the four most sensitive genera, all invertebrates, differed by only a factor of 2.6. Based on available data for saltwater organisms, the saltwater FAV was calculated to be 1.637μg/L. Following up the procedures identified in the EPA's *Guidance* (Stephan et al. 1985), CMC of 0.82 μg/L was calculated by dividing the FAV by 2.

The chronic toxicity of diazinon to saltwater organisms had been determined in a life-cycle test with the mysid, *A. bahia*, and a partial life-cycle test with the sheepshead minnow. The mysid test was of 22 days duration. There was no statistical difference in survival observed at any of the concentrations tested (0.54, 1.2, 2.1, 4.4 μg/L). The number of young per female was not significantly reduced relative to controls at diazinon concentrations < 2.1 μg/L. There were no young produced by females exposed to the highest concentration tested (4.4 μg/L). Based on reproduction, the chronic value for the mysid was the geometric mean of the chronic limits, 2.1 and 4.4 μg/L, or 3.040 μg/L. Dividing the acute value (4.82 μg/L) by the chronic value (3.040 μg/L) resulted in an ACR of 1.586 for the mysid, *A. bahia*. Sheepshead minnow reproduction was significantly reduced in all diazinon exposure concentrations during a partial life-cycle test. The number of eggs spawned per female in the 0.47, 0.98, 1.8, 3.5 and 6.5 μg/L diazinon concentrations were 69, 50, 50, 55 and 45 % of control fish, respectively. Neither survival nor growth was affected by diazinon exposures < 6.5 μg/L. Based on reduction of eggs spawned, the

chronic value for sheepshead minnow is < 0.47 μg/L. Dividing the acute value (1,400 μg/L) by the chronic value (< 0.47 μg/L) resulted in an ACR of > 2,979 for sheepshead minnow.

ACRs Chronic toxicity tests had been conducted on six aquatic species and chronic values ranged from 0.3882 μg/L for Ceriodaphnia, *C. dubia*, to 68.93 μg/L for flagfish. ACR for acutely sensitive crustacean invertebrates were 1.586 for mysids and 1.112 for *C. dubia*. In contrast, ratios were markedly higher for relatively acutely insensitive fishes: 23.84 for flagfish, 102.9 and 374.4 for fathead minnow > 903.8 for brook trout, and > 2,979 for sheepshead minnow. The *Guidelines* (Stephan et al. 1985) specify that if the SMACR seems to increase or decrease as the SMAV increases, the FACR should be calculated as the geometric mean of the ACRs for species whose SMAVs are close to the FAV. It did appear in this case that ACR values were lower for species acutely sensitive to diazinon and higher for acutely insensitive species. Therefore, only the acutely sensitive *C. dubia* and *A. bahia* ACRs were used to calculate the FACR of 1.328. The Guidelines also stipulate that if the most appropriate SMACRs are less than 2.0, acclimation has probably occurred during the chronic test and the FACR should be assumed to be 2.0. The low ACRs for *C. dubia* and *A bahia* support the finding that diazinon toxicity was rapid for these sensitive invertebrates and extended periods of exposure do not increase toxicity for these sensitive species. Thus, the FACR for diazinon was 2.0. The FCV for saltwater was calculated to be 0.8185 μg/L (FAVs divided by 2.0). Use of an FACR of 2.0 resulted in the same value for the CMC (acute criterion) and the CCC (chronic criterion).

The procedures described in the *Guidelines* (Stephan et al. 1985), indicate that saltwater aquatic organisms and their uses should not be affected unacceptably if the one-hour average concentration of diazinon does not exceed 0.82 μg/L more than once every three years on the average and if the four-day average concentration of diazinon does not exceed 0.82 μg/L more than once every three years on the average. The recommended exceedance frequency of 3 years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to diazinon exceeds the criterion. The USVI has adopted the above criteria for diazinon into the WQSR.

(g) Endosulfan (Alpha- and beta)

Endosulfan is an organochlorine insecticide that is being phased out globally. The USVI adopted alpha- and beta- endosulfan criteria of not to exceed 0.034 μg/L at any time (acute value) and 0.0087 μg/L chronic value for protection of aquatic life in all marine waters (class SA, SB and SC waters). These criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 1980 and described in EPA's Report: *Ambient Water Quality Criteria for Endosulfan* (EPA 1980g).

Data on acute toxicity of endosulfan were available for 12 saltwater fish and invertebrate species. Acute toxicity values ranged from 0.032 μg/L for a copepod to 730 μg/L for an annelid worm. The saltwater FAV for endosulfan, derived from the SMAV based on 12 species, was calculated to be 0.034 μg/L.

To address chronic toxicity, endosulfan 28-day life-cycle study was conducted with a saltwater mysid shrimp. In that study, the chronic limits were 0.33 and 0.71 μg/L; the geometric mean of these gives a chronic value of 0.48 μg/L. Both survival and reproduction (number of young per female) were affected at 0.71 μg/L, but not at 0.33 μg/L. The ACR for the species was 2.8. Sheepshead minnows were continuously exposed to endosulfan for 28 days starting with newly-fertilized eggs to the juvenile stage. Based on the results of this test, specifically the effects on growth of juvenile fish, the chronic limits were 0.27 and 0.6 μg/L, giving a chronic value of 0.40 μg/L. The ACR for the species was calculated to be 2.4. Overall, the ACRs for endosulfan ranged from 11 for Daphnia magna (the geometric mean of three values) to 2.4 for the sheepshead minnow. The resulting FACR was calculated to be 3.9. The saltwater FCV, obtained by dividing the FAV by the FACR, was calculated to be 0.0087 μg/L. These values are most appropriately applied to the sum of alpha-endosulfan and beta-endosulfan.

The resulting criterion for endosulfan to protect saltwater aquatic life is 0.0087 μg/L as a 24-hour average (chronic value), and the concentration which should not exceed of 0.034 μg//L at any time (acute value). The USVI has adopted the above criteria for endosulfan into the WQSR.

(h) Endrin,

Endrin is an organochloride which was primarily used as an insecticide, as well as a rodenticide and piscicide. The USVI adopted an endrin criteria of not to exceed 0.037 μg/L at any time (acute value) and 0.0023 μg/L chronic value for protection of aquatic life in all marine waters (class A, B and C waters). These criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 1980 and described in EPA's Report: *Ambient Water Quality Criteria for Endrin* (EPA 1980f).

Acute toxicity tests have been conducted with 17 species of saltwater fishes, and sensitivity varied from 0.048 μg/L for chinook salmon to 3.1 μg/L for northern puffer. Only two (usually tolerant) species, the sheepshead minnow and the sailfin molly have been tested for 96 hours in flow-through tests with measured endrin concentrations. Sheepshead minnow fry, juveniles, and adults did not differ in their sensitivity to acute exposure to endrin. Data on LC50 values for saltwater invertebrate species from acute toxicity tests on endrin supported the hypothesis that the acute toxicity of endrin was underestimated by static tests and by not measuring concentrations of endrin in test water. Acute values based on nominal concentrations for grass shrimp, and American oysters were higher than acute values for measured concentrations. Additionally, LC50 values based on static tests were greater than LC50 values for flow-through tests of the same duration for sheepshead minnow, sailfin mollies, shiner perch, dwarf perch, Korean shrimp, pink shrimp, and grass shrimp. The Saltwater FAV for endrin, derived from the SMAV was calculated to be 0.037 μg/L.

One saltwater invertebrate species, grass shrimp, has been exposed to endrin in a partial-life-cycle toxicity test. The number of females depositing embryos was less than that of controls, but embryo production and hatching success apparently were not affected. Larval mortality increased, time to metamorphosis increased, and growth of juvenile shrimp was decreased by

endrin concentrations of 0.11 μg/L and higher. A chronic value of 0.039 μg/L endrin was obtained for grass shrimp, even though all tested concentrations significantly impaired some life-cycle function. A lower limit of 0.03 μg/L was selected because the only effect was a delay in onset of spawning of about one week; a delay of one week probably would not affect natural populations. The upper limit of 0.05 μg/L was set based on decrease in number of ovigerous females and delay in spawning of 3 to 4 weeks. Sheepshead minnows, spot and mumnichog have been exposed to endrin for 10 days or longer. Of these tests with saltwater fish species, only the life-cycle exposure of sheepshead minnows was suitable for obtaining a chronic value (which was calculated to be 0.19 μg/L, making this species to be less sensitive than grass shrimp with FCV of 0.039 μg/L. For saltwater species, the ACRs for the sheepshead minnow and grass shrimp were calculated to be 1.9 and 18, respectively. Dividing the Saltwater FAV of 0.037 μg/L by the geometric mean of ACRs (4.0) gave the Saltwater FCV of 0.0093 μg/L.

The bioconcentraction of endrin from water into the tissues of saltwater organisms had also been well studied. Steady-state BCFs were available from studies with American oysters, grass shrimp, sheepshead minnows, and spot. Additional endrin BCF data were available from 96-hour exposures of oysters, grass shrimp, pink shrimp, sheepshead minnows, and sailfin mollies. BCFs for endrin in American oysters exposed for seven days ranged from 1,670 to 2,780. BCFs for endrin from two experiments with grass shrimp averaged 1,490 and 1,600. Average BCFs after a 96-hour exposure were 830 for grass shrimp and 980 for pink shrimp.

Bioconcentraction data for two of three species of saltwater fishes differ little from those for invertebrate species. Bioconcentraction factors calculated from nominal water concentrations were 1,340 for spot exposed for eight months and 1,560 for spot exposed five months. The average BCF for juvenile sheepshead minnows exposed for 28 days was 2,500; for adults exposed for 141 to 161 days the BCF was 6,400, and for juvenile exposed for four days the BCF was 2,600. Sailfin mollies exposed to endrin for four days had an average BCF of 2,400.

Dividing the FDA level by the geometric mean of normalized BCF (1,324) values and by a % lipid value of 16 for saltwater species, a saltwater residue value of 0.014 μg/L was calculated. Dividing the FDA action level of 0.3 mg/kg by the highest BCF for edible portion of an edible species, 2,780 for oyster provided an additional residue value for saltwater species of 0.11 μg/L. Dividing the FDA action level of 0.3 mg/kg for fish oil by the geometric mean of normalized BCF values (1,324) and by a % lipid value of 100 for fish oil gave a residue value for saltwater of 0.0023 μg/L. The lowest residue value of 0.0023 μg/L was taken as the Saltwater FRV. This value was lower than FCV of 0.0093 μg/L, resulting in the final chronic criterion for endrin based in the bioconcentration.

EPA's final recommendations stated that for endrin the criterion to protect saltwater aquatic life is 0.0023 μg/L (based on FRV) as a 24-hour average, and the concentration should not exceed 0.037 μg/L, at any time. The USVI adopted these criteria into the WQSR.

(i) Gamma BHC (Lindane)

Lindane is an organochlorine chemical that has been used both as an agricultural insecticide and as a pharmaceutical treatment for lice and scabies. In 2009, the production and agricultural use of lindane was banned under the Stockholm Convention on persistent organic pollutants.

The USVI adopted a lindane criterion of 0.16 µg/L (acute value) for protection of saltwater aquatic life in all marine waters (class A, B and C waters). This criterion was adopted based on EPA's recommended 304(a) aquatic life criterion published in 1980 and described in EPA's Report: *Ambient Water Quality Criteria for Hexachlorocyclohexane* (EPA 1980d).

Acute toxicity values for gamma- benzenehexachloride (gamma-BHC) [also known as gamma-hexachlorocyclohexane (gamma-HCH) or lindane] with saltwater invertebrate species ranged from 0.17 to 3,680 µg/L. Saltwater invertebrate species were generally more sensitive than fish species to lindane. The LC50 for the commercially important pink shrimp was more than one order of magnitude lower than the second most sensitive species. The least sensitive invertebrate species was the potychaete, with a 96 hour LC50 value of 3,680 µg/L (21,000 times greater than that of the pink shrimp).

Although saltwater fish species have a wide range of sensitivity to lindane, they are generally less sensitive than saltwater invertebrate species. Eleven species of fishes were tested in static and flowthrough exposures. Only two species were exposed for 96 hours under flow-through conditions with measured concentrations. These LC50 values were 30.6 µg/L for the pinfish and 103.9 µg/L for the sheepshead minnow. LC50 values, including nine other species, had a range from 7.3 to 103.9 µg/L. The Saltwater FAV for lindane (derived from the SMAV) was calculated to be 0.16 µg/L. No chronic toxicity values for HCH were found for any saltwater invertebrate or fish species.

The Saltwater FAV for lindane of 0.16 µg/L was adopted by the USVI, as an acute value, into the WQSR.

(j) Guthion

Guthion (azinophos-methyl) is a broad spectrum organophosphate insecticide. The USVI adopted a guthion criterion of 0.01 µg/L (chronic value) for protection of saltwater aquatic life in all marine waters (class A, B and C waters). This criterion was adopted based on EPA's recommended 304(a) aquatic life criterion published in 1976 and described in EPA's "Red Book" Report: *Quality Criteria for Water* (EPA 1976).

Ninety-six-hour LC50 values obtained for aquatic invertebrates exposed to guthion ranged from 0.10 to 22.0 µg/L. The grass shrimp was exposed to guthion in a continuous flow bioassay for up to 20 days and it was found that the 5- and 20-day LC50 values were 1.2 and 0.16 µg/L, respectively. Studies found that the amphipod was the most sensitive aquatic organism tested, with a 96-hour LC50 of 0.10 µg/L. Ninety-six-hour LC50 values for fish exposed to guthion

ranged from 4 to 4270 μg/L. An estimated "safe" long-term exposure concentration for fathead minnows lied between 0.3 and 0.5 μg/L.

Organophosphate pesticides (including guthion) are toxic because they inhibit the AChE, which is essential to nerve impulse conduction and transmission. Studies demonstrated that a 40 to 70 % inhibition of fish brain AChE usually is lethal. Centrarchids generally are considered one of the more sensitive groups of fish to guthion. Studies found that over a 15- day period, bluegills exhibited AChE inhibition at 1.0 μg/L guthion, but not at 0.1 μg/L.

The 24-hour LC50 for the white mullet was found to be 5.5 μg/L guthion. The 96- hour LC50 for the striped mullet was determined to be *8* μg/L guthion. The 48-hour LC50 for the fish, *Pleuronectesli manda*, was reported to be 10 to 30 μg/L. The 48-hour LC50 for the European shrimp was found to be 0.33 μg/L guthion. Studies found that the 24-hour EC50 for blue crab was 550 μg/L and the 48-hour EC50 for pink shrimp was 4.4 μg/L guthion.

A criterion level of 0.01 μg/L for guthion was derived based upon use of 0.1 application factor applied to the 96-hour LC50 of 0.1 μg/L for Gammarus and a similar value of 0.33 μg/L for the European shrimp. The use of an application factor of 0.1, in this derivation process, was consistent with the EPA's recommendation to provide an additional degree of protection for all life stages of the test organism in water of varying quality, as well as to protect associated organisms within the aquatic environment that have not been tested and that may be more sensitive to the test constituent (EPA 1976).

In summary, the EPA's 304(a) recommendation is that, for guthion, the criterion to protect saltwater aquatic life is 0.01 μg/L, as a chronic value. The USVI has adopted this criterion for guthion into the WQSR.

(k) Heptachlor and Heptachlor Epoxide

Heptachlor is an organochlorine compound that was used as an insecticide. The US EPA has limited the sale of heptachlor products to the specific application of fire ant control in underground transformers. The USVI adopted a heptachlor criteria of 0.053 μg/L (acute value) and 0.0036 μg/L chronic value for protection of saltwater aquatic life in all marine waters (class A, B and C waters). The same criteria values have been adopted by the USVI for Heptachlor Epoxide. These criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 1980 and described in EPA's Report: *Ambient Water Quality Criteria for Heptachlor* (EPA 1980c).

In general, heptachlor is quite stable to chemical reactions, however, in the environment, heptachlor undergoes numerous microbial, biochemical, and photochemical reactions.

Conversion of heptachlor to heptachlor epoxide has been reported in microorganisms, plants, soils and in mammals. Heptachlor epoxide represents the principal metabolite of heptachlor. The persistence of both, heptachlor and heptachlor epoxide in the environment is well-known.

Both heptachlor and heptachlor epoxide had been reported in fish residues. There were few data on the relative toxicity to aquatic organisms of these two materials. What data were available suggest that the toxicity of epoxide is similar to the toxicity heptachlor itself. There were insufficient saltwater data to evaluate relative toxicity of heptachor and heptachlor epoxide.

Heptachlor had been shown to be acutely toxic to saltwater fish and invertebrate species. Saltwater invertebrate species seemed to be more sensitive than fish species to heptachlor and heptachlor epoxide and demonstrate a greater variability in sensitivity between species. Of the seven species tested, the commercially valuable pink shrimp was especially sensitive with 96-hour LC50 values as low as 0.03 μg/L. Other species, such as the blue crab and American oyster, were 2,100 to 950 times less sensitive, respectively, than the pink shrimp.

Ninety-six hour LC50 of heptachlor for the grass shrimp, based on a static exposure, was 440 μg/L, whereas the result from a flow-through test with measured concentrations was 1.06 μg/L. A similar relationship was true for the American oyster. Test results from a flow-through exposure with were 27 - 30 μg/L and, using flow-through procedures and measured concentrations, a 96-hour LC50 was determined to be 1.5 μg/L. Generally, toxicity data obtained from static tests or those in which concentrations were not measured yielded higher acute values for heptachlor than other tests. The range of LC50 values for saltwater invertebrate species was calculated from 0.03 to 440 μg/L. The 96-hour LC50 values derived from flow-through tests with four saltwater fish species ranged from 0.85 to 10.5 μg/L. Results of static exposures of eight fish species were variable and showed higher LC50 values than those from flow-through tests. The Saltwater FAV for heptachlor, derived from the SMAV, was calculated to be 0.053 μg/L.

A 28-day life-cycle toxicity test was completed with a saltwater mysid shrimp, which was exposed to measured concentrations of 0.17, 0.64, 1.3, and 3.1 μg/L. Cumulative mortality of test animals exposed to 0.64 μg/L of heptachlor was the most sensitive effect. The chronic toxicity of technical heptachlor to the sheepshead minnow was measured in an l-week partial life-cycle exposure begun with juveniles. Survival was affected at concentrations of 2.8 μg/L and greater. Embryo production was significantly decreased at the lowest concentration tested, 0.71, and at test concentrations of 1.9 to 5.7 μg/L. The chronic toxicity of technical heptachlor to sheepshead minnows was also measured in a separate 28-day early life-stage test. Hatching was unaffected, but survival of fry was significantly reduced from that of controls at measured concentrations of 2.24 to 4.3 μg/L. Comparison of these data with that from the early life-stage portion of the partial life-cycle exposure shows survival of fry was reduced at a similar concentration in both exposures (2.24 and 2.8 μg/L, respectively). Growth of fry in the early life-stage test was significantly reduced at concentrations of 2.04 μg/L and above. Chronic values for saltwater species were be obtained from only the sheepshead minnow early life-stage test and not the life-cycle tests on this fish species and mysid shrimp. The chronic value from the early life-stage test was 1.58 μg/L, and the ACR was 3.9.

The saltwater bioconcentration data showed that uptake of heptachlor was fairly rapid, reaching a maximum in one study in three days. However, heptachlor was readily metabolized in fish to heptachlor epoxide. The relative amount of heptachlor epoxide in tissues increased with length of exposure, with the maximum amount occurring by day 17. After a 28-day deputation, approximately 90 % of the heptachlor was either eliminated or degraded to heptachlor epoxide.

From available data on the bioconcentration of heptachlor and heptachlor epoxide from water into the tissues of saltwater organisms, the only BCF values available at steady-state for heptachlor and heptachlor epoxide were those for fish species. Spot exposed for 24 days to technical grade material reached a maximum concentration of heptachlor in whole body after three days. In the same exposure, maximum levels of heptachlor epoxide were reached in whole fish after 17 days. Juvenile sheepshead minnows exposed in two separate expert 3 for 28 days to technical grade material had similar BCF values (4,667 and 5,700). Adult sheepshead minnows exposed to technical grade material for 126 days accumulated heptachlor and heptachlor epoxide to a much greater extent, an average 37,000 times that in the exposure water. The BCF values derived in the above studies are from effect, as well as safe concentrations, and they appear similar. The only BCF values considered appropriate for heptachlor for the derivation of a FRV were those based on the concentration of heptachlor in water and the total concentration of heptachlor and heptachlor epoxide in tissue. Dividing the FDA action level of 0.3 mg/kg by the geometric mean of normalized BCF values (5,222) and by a % lipid value of 16 for saltwater species gave a saltwater FRV of 0.0036 µg/L.

For both, heptachlor and heptachlor epoxide, the EPA recommended criteria to protect saltwater aquatic life of 0.0036 µg/L (derived based on FRV) as a 24-hour average, and the concentration of not to exceed 0.053 µg/L at any time. The USVI adopted these criteria into the WQSR.

(l) Malathion

Malathion is an insecticide. The USVI adopted a malathion criterion of 0.1 µg/L (chronic value) for protection of saltwater aquatic life in all marine waters (class A, B and C waters). This criterion was adopted based on EPA's recommended 304(a) aquatic life criterion published in 1976 and described in EPA's "Red Book" Report: *Quality Criteria for Water* (EPA 1976).

Toxicity studies for malathion have been completed on a number of marine animals. Reported 96-hour LC50 values were as follows: 125 µg/L for *Menidia menidia*, 550 µg for *Mugil cephalus*, 250 µg for *Fundulus rnnialis*, 240 µg for *Fundulus heteroclitus*, and 27 µg for *Thalassoma bifasciatum*. Studies of the toxicity of malathion on marine invertebrates reported the 96-hour LC50 to be 33 µg/L for sand shrimp, 82 µg/L for grass shrimp, and 83 µg/L for hermit crab. Growth of oyster, *Crassostrea virginica*, was reduced 32 % by 96-hour exposure to 1 mg/L. The 48- hour LC50 for fertilized eggs of oysters was estimated to be 9.07 mg/L and the 14-day LC50 for larvae, 2.66 mg/L.

As it is specified in the "Red Book" (EPA 1976), in cases where only 96-hour bioassay data are available, EPA recommends the application of the substantial safety factor to protect all life stages of the test organism in water of varying quality, as well as to protect associated organisms within the aquatic environment that have not been tested and that may be more sensitive to the test constituent. In such cases, application factors are to be used to provide the degree of protection required. Safe levels for certain chlorinated hydrocarbons and certain heavy metals were estimated by EPA by applying 0.01 application factor to the 96-hour LC50 value for sensitive aquatic organisms.

The above described EPA recommendation was applied in the process of derivation of marine criterion for malathion. In general, many aquatic invertebrates appear to be more sensitive than fish to malathion. The 96-hour LC50 for *Gammarus lacustris* was 1.0 µg/L, for *G. fasciatis*, 0.76 µg/L and Daphnia 1.8 µg/L.

Malathion enters the aquatic environment primarily as a result of its application as an insecticide. Because it degrades quite rapidly in most waters, depending on pH, its occurrence is sporadic rather than continuous. Because the toxicity is exerted through inhibition of AChE and because such inhibition may be additive with repeated exposures and may be caused by any of the organophosphorus insecticides, inhibition of AChE by more than 35 % may be expected to result in damage to aquatic organisms.

An application factor of 0.1 was applied to the 96-hour LC50 data for *Gammarus lacustris, G. fasciatis* and Daphnia, which were all approximately 1.0 µg/L, yield a criterion of 0.1 µg/L. The use of an application factor of 0.1, in this derivation process, was consistent with the EPA's recommendation to provide an additional degree of protection for all life stages of the test organism in water of varying quality, as well as to protect associated organisms within the aquatic environment that have not been tested and that may be more sensitive to the test constituent (EPA 1976).

In summary, the EPA's 304(a) recommendation is that, for malathion, the criterion to protect saltwater aquatic life is 0.1 µg/L (chronic value). The USVI has adopted this criterion for malathion into the WQSR.

(m) Methoxychlor

Methoxychlor is a synthetic organochlorine used as an insecticide..The USVI adopted a methoxychlor criterion of 0.03 µg/L (chronic value) for protection of saltwater aquatic life in all marine waters (class A, B and C waters). This criterion was adopted based on EPA's recommended 304(a) aquatic life criterion published in 1976 and described in EPA's "Red Book" Report: *Quality Criteria for Water* (EPA 1976).

As it is specified in the "Red Book" (EPA 1976), in cases where only 96-hour bioassay data are available, EPA recommends the application of the substantial safety factor to protect all life stages of the test organism in water of varying quality, as well as to protect associated organisms within the aquatic environment that have not been tested and that may be more sensitive to the test constituent. In such cases, application factors are to be used to provide the degree of protection required. Safe levels for certain chlorinated hydrocarbons and certain heavy metals were estimated by EPA by applying 0.01 application factor to the 96-hour LC50 value for sensitive aquatic organisms.

The above described EPA recommendation was applied in the process of derivation of marine criterion for methoxychlor. Only few studies have been completed to address impact of the methoxychlor on marine species. The 96-hour LC50 of methoxychlor for the pink shrimp was

reported to be 3.5 µg/L and the 30-day LC50 was reported to be 1.3 µg/L. By using an application factor of 0.01 with the pink shrimp's acute toxicity of 3.5 µg/L, the recommended criterion for the marine environment is 0.03 µg/L. The use of an application factor of 0.1, in this derivation process, was consistent with the EPA's recommendation to provide an additional degree of protection for all life stages of the test organism in water of varying quality, as well as to protect associated organisms within the aquatic environment that have not been tested and that may be more sensitive to the test constituent (EPA 440/9-76-023).

The BCF of 470 and 1,500 was calculated for two mollusks, when exposed to 1 µg/L of metoxychlor for five days. Using the 1,500 BCF as basis, a water concentration of 0.2 µg/L would be required to meet the U.S. Food and Drug Administration's guideline for metoxychlor in animal tissue. Thus, the recommended marine criterion of 0.03 µg/L is an order of magnitude lower than this concentration. The USVI has adopted the criterion for methoxychlor of 0.03 µg/L, as chronic value, into the WQSR.

(n) Mirex,

Mirex is an organochloride insecticide. The USVI adopted a mirex criterion of 0.001 µg/L (chronic value) for protection of saltwater aquatic life in all marine waters (class A, B and C waters). This criterion was adopted based on EPA's recommended 304(a) aquatic life criterion published in 1976 and described in EPA's "Red Book" Report: *Quality Criteria for Water* (EPA 1976).

Data upon which to base a marine criterion for mirex involve several estuarine and marine crustaceans. A concentration of 0.1 µg/L technical grade mirex in flowing seawater was reported to be lethal to juvenile pink shrimp, *Penaeus durorarum*, in a 3-week exposure. In static tests with larval stages of the mud crab, *Rhithropanopeus harrisii*, reduced survival was observed in water containing 0.l µg/L concentration of mirex. In three of four 28-day seasonal flow-through experiments, studies found reduced survival of *Penaeus durorarum*, and grass shrimp, *Palaemonetes pugio*, at levels of 0.12 µg/L mirex in summer, 0.06 µg/L mirex in fall, and 0.09 µg/L mirex in winter. Studies reported that effects of mirex on estuarine and marine crustaceans were observed only after considerable time had elapsed. Thus, it seems reasonable that length of exposure is an important consideration for this chemical. By using the application factor of 0.01 to a reasonable average of toxic-effect levels for mirex, as summarized above, a recommended marine criterion for mirex results in the concentration of 0.001 µg/L. The use of an application factor of 0.1, in this derivation process, was consistent with the EPA's recommendation to provide an additional degree of protection for all life stages of the test organism in water of varying quality, as well as to protect associated organisms within the aquatic environment that have not been tested and that may be more sensitive to the test constituent (EPA 1976).

The USVI has adopted the above derived criterion for mirex into the WQSR.

(o) Pentachlorophenol

Pentachlorophenol (PCP) is an organochlorine compound used as a pesticide and a disinfectant. The USVI adopted a PCP criteria of 13 μg/L (acute value) and 7.9 μg/L (chronic value) for protection of saltwater aquatic life in all marine waters (class A, B and C waters). These criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 1995 in the Report: *Ambient Water Quality Criteria for Pentachlorophenol* (EPA 1995). These criteria for PCP was derived using the *Guidelines* (Stephan et al. 1985).

Acute toxicity values from tests with 18 species of saltwater animals, representing 17 genera, ranged from 63 μg/L for late yolk-sac larvae of the Pacific herring, to 18,000 μg/L for adult blue mussels. The embryo and larval stages of invertebrates and the late larval premetamorphosis stage of ash appeared to be the most sensitive life stages to PCP. Salinity, temperature, and pH had a slight effect on the toxicity of PCP to some saltwater animals.

Life-cycle toxicity tests had been conducted with the sheepshead minnow. The chronic value for the minnow was 64.31 μg/L and the ACR was 6.873. The EC50 for saltwater plants ranged from 17.4 (for the diatom) to 3,600 (for green algae). Apparent steady-state BCFs were available for the eastern oyster and two saltwater fishes and range from 10 to 82.

The procedures described in the *Guidelines* indicated that saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of pentachlorophenol does not exceed 7.9 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 13 μg/L more than once every three years on the average. The recommended exceedance frequency of 3 years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to PCP exceeds the criterion.

The USVI has adopted the above criteria for PCP into the WQSR.

(p) Toxaphene,

Toxaphene was an insecticide used primarily for cotton in the southern United States during the late 1960s and 1970s. Toxaphene was banned in the United States in 1990 and was banned globally by the 2001 Stockholm Convention on Persistent Organic Pollutants. The USVI adopted a toxaphene criteria of 0.21 μg/L (acute value) and 0.0002 μg/L, chronic value for protection of saltwater aquatic life in all marine waters (class A, B and C waters). These criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 1986 and described in EPA's Report: *Ambient Aquatic Life Water Quality Criteria for Toxaphene* (EPA 1986b). These criteria for toxaphene were derived using the *Guidelines* (Stephan et al. 1985).

Acute toxicity values for saltwater animals are obtained from tests with nine intervertebrate and six dish species. The sensitivities of the tested species ranged from 0.53 μg/L for juvenile pinfish to 460,000 μg/L for adult of the clam, Rangia cuneate. Acute values for stage II and III

larvae of the drift line crab were 0.5542 μg/L and 0.5298 μg/L, respectively, which are similar to the acute value for the pinfish. In general, fishes and invertebrates were found to be similarly sensitive with acute values ranging from 0.53 to 31.32.

Of the fifteen saltwater genera for which acute values were available, the most sensitive, Lagodon, was over 867,000 times more sensitive than the most resistant, Rangia. The four most sensitive genera included three fishes and an invertebrate. The saltwater FAV was calculated to be 0.4197 μg/L, which is below the acute value for the most sensitive species. The FAV divided by two resulted in the final acute criterion of 0.21 μg/L.

The chronic toxicity tests that have been conducted with saltwater species included an early-stage test and a life-cycle test with the sheepshead minnow, an early life-stage test with longnose killifish, and a life-cycle test with the mysid. Early life-stage toxicity tests have been conducted with the sheepshead minnow, *Cmrinodon varieqatus*, and the longnose killifish, *Fundulus similis*, whereas life-cycle tests have been conducted with the sheepshead minnow and a mysid. For the sheepshead minnow, chronic values of 1.658 μg/L from the early life-stage test and 0.7141 μg/L from the life-cycle toxicity test are similar to the 96-hr LC50 of 1.1 μg/L. Killifish are more chronically sensitive with effects noted at 0.3 μg/L. In the life-cycle test with the mysid, no adverse effects were observed at the highest concentration tested, which was only slightly below the 96-hr LC50, resulting in an ACR of 1.132.

The procedures described in the *Guidelines* indicate that saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of toxaphene does not exceed 0.0002 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.21 μg/L more than once every three years on the average. The recommended exceedance frequency of 3 years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to toxaphene exceeds the criterion.

The USVI has adopted these criteria for toxaphene into the WQSR.

(q) 4, 4'DDT

DDT (dichlorodiphenyltrichloroethane) is an organochloride known for its insecticidal properties. The USVI adopted a 4, 4' DDT criteria of 0.13 μg/L (acute value) and 0.001 μg/L chronic value for protection of saltwater aquatic life in all marine waters (class A, B and C waters). These criteria apply to DDT and its metabolites. The acute criterion was adopted based on EPA's recommended 304(a) aquatic life criterion published in 1980 and described in EPA's Report: *Ambient Water Quality Criteria for 4, 4'DDT* (EPA 1980a).

Acute toxicity tests on six saltwater invertebrate species produced acute LC50 values from 0.14 to 9.0 μg/L, with the lowest value was the 96-hour LC50 for the brown shrimp. Data were available for a mollusc and four different families of arthropods. Both, 24- and 48-hour values for five species giving EC50 ranged from 0.6 to 10 μg/L. Acute 96-hour toxicity tests with 11 species of saltwater fishes (representing nine fish families) gave LC50 values ranging from 0.26

to 89 μg/L. The northern puffer was by far the least sensitive; most other LC50 values for fish species ranged between 0.5 and 7 μg/L. The 48-hour LC50 values for six species ranged from 0.32 to 3.2 μg/L. The Saltwater FAV for DDT, derived from the SMAVs, using the procedure described in the Guidelines, was calculated to be 0.13 μg/L.

No chronic toxicity data were found for any saltwater animal species.

BCFs were available for three saltwater invertebrate and nine fish species. BCFs from laboratory tests with DDT and saltwater organisms ranged from 1,200 to 76,300 for fish and shellfish.

Eastern oysters provided BCF values from 42,400 in a 252-day exposure to 76,300 in a 168-day exposure. BCF values in these studies ranged from 4,750 for Cancer magister to 46,500 for the dwarf perch. Dividing a BCF value by the % lipid value for the same species provided a BCF value adjusted to 1 % lipid content (this resultant BCF value is referred to as the normalized BCF). The geometric mean of normalized BCF for DDT for freshwater and saltwater aquatic life was calculated to be 17,870.

Dividing the FDA action level of 5.0 mg/kg for fish by the geometric mean of normalized BCF values (17,870) and by a % lipid value of 15 for freshwater species gave a freshwater residue value based on marketability for human consumption of 0.019 μg/L. Dividing the FDA action level (5.0 mg/kg) by the geometric mean of normalized BCF values (17,870) and by a % lipid value of 16 for saltwater species gave a saltwater residue value of 0.017 μg/L. Also based on marketability for human consumption, using the FDA action level and the highest BCF for edible portion of a consumed fish species (458,259 for lake trout freshwater), a residue value of 0.011 μg/L was obtained for freshwater. No appropriate BCF value for edible portion of a consumed fish species was available for saltwater.

A residue value for wildlife protection of 0.001 μg/L was obtained for both freshwater and saltwater using the lowest maximum permissible tissue concentration of 0.15 mg/kg based on reduced productivity of the brown pelican.

Average lipid content of pelican diets was unavailable. Clupeids usually constitute the major prey of pelicans, and the % lipid value of the clupeid, northern anchovy, is 8. The northern anchovy is in some areas a major food source of the brown pelican. Therefore, the % lipid value of 8 was used for the calculation of the FRV. The value of 0.15 mg/kg divided by the geometric mean of normalized BCF values (17,870) and by a % lipid value of 8 gave a residue value of 0.001 μg/L. Selection of the lowest freshwater and saltwater residue values from the above calculations gave a Freshwater FRV of 0.001 μg/L and a Saltwater FRV of 0.0010 μg/L.

For dichlorodiphenyltrichloroethane (DDT) and its metabolites the criterion to protect saltwater aquatic life was derived using the *Guidelines* is 0.0010 μg/L as a 24-hour average and the concentration should not exceed 0.13 μg/L at, at any time. The USVI adopted these criteria into the VIWQSR.

(2) Nonylphenol

Nonylphenols are a family of closely related organic compounds used in manufacturing of antioxidants, detergents, paints, pesticides and plastics. The USVI adopted a nonylphenol criteria of 7 μg/L (acute value) and 1.7 μg/L chronic value for protection of saltwater aquatic life in all marine waters (class A, B and C waters). Both criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 2005 and described in EPA's Report: *Aquatic Life Ambient Water Quality Criteria for Nonylphenol - Final* (EPA 2005a). The criterion for nonylphenol was derived using the *Guidelines* (Stephan et al. 1985).

Acute toxicity of nonylphenol was tested in 11 saltwater species (two fish and nine invertebrates), from 11 genera. The SMAV ranged from 17 μg/L for the winter flounder to 209.8 μg/L for the sheepshead minnow. The saltwater FAV was calculated to be 13.93 μg/L, which divided by two resulted in the acute value of 7 μg/L.

Chronic toxicity of nonylphenol was tested on one saltwater species, the mysid, which was also the most sensitive of all species tested, both freshwater and saltwater. Two tests were available with chronic values of 5.112 μg/L (reduced growth) and 12.02 μg/L (reproductive endpoint). Based on growth, the NOEC and LOEC determined in this study were 3.9 and 6.7 μg/L, respectively. The chronic value calculated as the geometric mean of the NOEC and LOEC, was 5.112 μg/L. Dividing the acute value of 43 by the chronic value of 5.112 μg/L resulted in the ACR of 8.412.

The procedures described in the *Guidelines* indicate that saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of nonylphenol does not exceed 1.7 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 7.0 μg/L more than once every three years on the average. The recommended exceedance frequency of 3 years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to toxaphene exceeds the criterion. The USVI has adopted these criteria for toxaphene into the WQSR.

(3) Tributyltin (TBT)

Tributyltin (TBT) is the active ingredient of many products that act as biocides against a broad range of organisms. It is primarily used as an antifoulant paint additive on ship hulls, docks, fishnets, and buoys to discourage the growth of marine organisms such as bacteria, mussels or algae. The USVI adopted a tributyltin criteria of 0.42 μg/L (acute value) and 0.0074 μg/L chronic value for protection of saltwater aquatic life in all marine waters (class A, B and C waters). Both criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 2003 and described in EPA's Report: *Ambient Aquatic Life Water Quality Criteria for Tributyltin (TBT) - Final* (EPA 2003). These criteria for nonylphenol were derived using the *Guidelines* (Stephan et al. 1985).

A partial life-cycle TBT test of one-year duration was conducted with the snail, *Nucella lapillus*. The chronic value for this species was 0.0143 μg/L. A life-cycle test was conducted with the

copepod, *Eurytemora affinis*. The chronic value for this test was 0.145 μg/L and the acute/chronic ratio was calculated to be 15.17. A life-cycle toxicity test was conducted with the saltwater mysid, *Acanthomysis sculpta*. The chronic value was 0.1308 μg/L based on reduced reproduction and the ACR was 4.664.

Tributyltin chronically affects certain saltwater copepods, gastropods, and pelecypods at concentrations less than those predicted from "standard" acute and chronic toxicity tests. The data demonstrated that reductions in growth occur in commercially or ecologically important saltwater species at concentrations of TBT less than the FCV of 0.0658 μg/L derived using FAVs and ACRs. Survival of the copepod *A. tonsa* was reduced in 0.023 μg/L. Growth of larvae or spat of two species of oysters, *Crassostrea gigas* and *Ostrea edulis* was reduced in about 0.02 μg/L; some *C. gigas* larvae died in 0.025 μg/L. Shell thickening and reduced meat weights were observed in *C. gigas* at 0.01 μg/L. Reproductive effects were observed with *N. lapillus* at TBT concentrations >0.0074 μg/L. Since these levels were ones at which an effect was seen, a protective level for these commercially important species is, therefore, below 0.01 μg/L.

Tests of the acute toxicity of TBT to resident North American saltwater species that are useful for deriving water quality criteria concentrations have been performed with 26 species of invertebrates and seven species of fish. The range of acute toxicity to saltwater animals is a factor of about 1,176. Acute values range from 0.24 μg/L for juveniles of the copepod, *Acartia tonsa* to 282.2 μg/L for adult Pacific oysters. The 96-hr LC50s for seven saltwater fish species range from 1.460 μg/L for juvenile chinook salmon, to 25.9 μg/L for sub adult sheepshead minnows. Larval bivalve molluscs and juvenile crustaceans appear to be much more sensitive than adults during acute exposures. The 96-hr LC50 for larval Pacific oysters was 1.557 μg/L, whereas the value for adults was 282.2 μg/L. The 96-hr LC50s for larval and adult blue mussels were 2.238 and 36.98 μg/L, respectively. Juveniles of the crustacean *Acanthomysis sculpta* were slightly more sensitive to TBT than adults. The GMAV for 30 saltwater genera range from 0.61 μg/L for *Acanthomysis* to 204.4 μg/L for *Ostrea*. The GMAV for the 10 most sensitive genera differ by a factor of less than four. Included within these genera are four species of molluscs, six species of crustaceans, and one species of fish. The saltwater FAV for TBT was calculated to be 0.8350 μg/L, which is greater than the lowest saltwater SMAV of 0.61 μg/L. The saltwater CMC is 0.4175 μg/L and is calculated by dividing the FAV by two.

Life-cycle toxicity tests were conducted with the saltwater mysid, *Acanthomysis sculpta.* The effects of TBT on survival, growth, and reproduction of *A. sculpta* were determined in five separate tests lasting from 28 to 63 days. The tests separately examined effects of TBT on survival (1 test), growth (3 tests) and reproduction (1 test) instead of the approach of examining all endpoints in one life-cycle test. All tests began with newly released juveniles and lasted through maturation and spawning; therefore, they are treated as one life-cycle test. The number of juveniles released per female at a TBT concentration of 0.19 μg/L was 50% of the number released in the control treatment, whereas the number released at the next lower TBT concentration (0.09 μg/L) was not significantly different from the control treatment. Reductions in the number of juveniles released resulted from deaths of embryos within brood pouches of individual females and not from reduced fecundity. Numbers of females releasing viable juveniles was reduced in 0.19 and 0.33 μg/L. At concentrations of 0.38 μg/L and above, survival and weight of female mysids were reduced; all mysids in 0.48 μg/L died. The chronic value

(0.1308 μg/L) is the geometric mean of 0.09 μg/L and 0.19 μg/L and is based upon reproductive effects. The ACR is 4.664 when an acute value of 0.61 μg/L is used.

The Final ACR of 12.69 was calculated as the geometric mean of the ACRs of 36.60 for *D. magna*, 10.01 for *P. promelas*, 4.664 for *A. sculpta* and 15.17 for *E. affinis*. Division of the saltwater FAV by 12.69 results in FCV for saltwater of 0.0658 μg/L. The National Guidelines (Stephan et al. 1985) require that the criterion be lowered if sound scientific evidence indicates that adverse effects might be expected on important species. The above data demonstrate that the reductions in growth occur in commercially or ecologically important saltwater species at concentrations of TBT less than the FCV of 0.0658 μg/L derived using FAV and ACRs. Consistent with the Guidelines directive to consider other relevant data when establishing criteria, EPA believes the FCV should be lowered to 0.0074 μg/L.

The procedures described in the *Guidelines* indicate that, for TBT, the criterion to protect saltwater does not exceed 0.42 μg/L more than once every three years on the average (acute criterion) and the four-day average concentration of TBT does not exceed 0.0074 μg/L more than once every three years on the average (chronic criterion). The USVI adopted these criteria into WQSR.

(4) Polychlorinated biphenyls (PCBs),

A polychlorinated biphenyl (PCB) is a synthetic organic chemical compound used as dielectric and coolant fluids. PCB production was banned by the United States Congress in 1979 and by the Stockholm Convention on Persistent Organic Pollutants in 2001.

The USVI adopted a PCBs criterion of 0.03 μg/L chronic value for protection of saltwater aquatic life in all marine waters (class A, B and C waters). This criterion applies to total PCBs (e.g., the sum of all congener or all isomer or homolog analyses. This criterion was adopted based on EPA's recommended 304(a) aquatic life criterion published in 1980 and described in EPA's Report*: Ambient Water Quality Criteria for Polychlorinated Biphenyls* (EPA 1980i).

SMAV for PCBs and saltwater animals ranged from 10.5 μg/L to 20 μg/L from six tests on three invertebrate species. The LC50 values for saltwater invertebrate species ranged from 10.2 μg/L to 60 μg/L No chronic tests have been reported, in which saltwater invertebrate species were exposed to PCBs. Two chronic tests were conducted on the sheepshead minnow, providing chronic values for this species of 7.14 μg/L and 0.098 μg/L. The saltwater FRV was calculated to be 0.03 μg/L.

For PCBs the criterion to protect saltwater aquatic life as derived using the Guidelines is 0.030 μg/L as a 24-hour average. The available data indicate that acute toxicity to saltwater aquatic life probably will only occur at concentrations above 10 μg/L and that the 24-hour average criterion should provide adequate protection against acute toxicity. The USVI has adopted this criterion for PCBs into the WQSR.

b) Inorganic Compounds

(1) Metals

(a) Arsenic

The USVI adopted an arsenic criteria of 69 µg/L (acute value) and 36 µg/L (chronic value), expressed in terms of the dissolved metal, for protection of saltwater aquatic life in all marine waters (class A, B and C waters). The criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 1985 and described in EPA's Report: *Ambient Water Quality Criteria for Arsenic – 1984*.

This recommended water quality criterion for arsenic was derived from data for arsenic (III), but is applied to total arsenic. This might imply that arsenic (III) and arsenic (V) are equally toxic to aquatic life and that their toxicities are additive. No data are known to be available concerning whether the toxicities of the forms of arsenic to aquatic organisms are additive.

Data were available on the acute toxicity of inorganic arsenic (III) to saltwater species in three fish and eight invertebrate genera. The fish species tested were the most resistant, with a range of LC50s from 12,700 µg/L for the sheepshead minnow to 16,030 µg/L for the Atlantic silverside. Among the invertebrates, the lowest acute value, 232 µg/L, was obtained with zoeae of the Dungeness crab whereas the highest value, 10,120 µg/L, was from a test with the polychaece worm, *Neanches arenaceodencaca*. Of the eleven GMAVs, all eight for invertebrates were lower than the three for fish. The saltwater FCV for inorganic arsenic (III) was 137.1 µg/L, which was about one-half the lowest SMAV. Data were available for inorganic arsenic (V) with two saltwater species. Arsenic (V) was more toxic than arsenic (III) to the amphipod, *Ampelisca abdita*, which SMAVs were 4,610 µg/L for arsenic (V) and 8,227 µg/L for arsenic (III). Not enough data were available to calculate a saltwater FAV for inorganic arsenic (V).

The chemistry of arsenic in water is complex and the form present in solution is dependent on such environmental conditions as pH, organic content, suspended solids, and sediment. The relative toxicities of the various forms of arsenic apparently vary from species to species. Twelve species of saltwater animals have acute values for inorganic arsenic (III) from 232 to 16,030 µg/L and the single ACR was calculated to be 1.945. The only acute values available for inorganic arsenic (V) were for two invertebrate and were between 2,000 and 3,000 µg/L. Arsenic (III) and arsenic (V) were equally toxic to various species of saltwater algae, but the sensitivities of the species ranged from 19 µg/L to more than 1,000 µg/L.

Data on the chronic toxicity of arsenic to saltwater species were available for only one species, *Mysidoosis bahia*. In a 35-day life-cycle test on arsenic (III), no adverse effects were statistically significant at 631 µg/L, whereas 1,270 µg/L affected reproduction and significantly reduced survival. These results provided a chronic value of 895.2 µg/L and an ACR of 1.944 µg/L.

The four ACRs available for inorganic arsenic (III) were 4.748, 4.660, 4.862, and 1.944 and the geometric mean of 3.803 was the Final ACR. Division of the saltwater FAV by this ratio resulted in saltwater FCV of 36.05 μg /L.

Very few data were available concerning the toxicity of any form of arsenic other than inorganic arsenic (III) to saltwater aquatic life. This recommended water quality criterion was derived from data for arsenic (III), but is applied here to total arsenic, which might imply that arsenic (III) and arsenic (V) are equally toxic to aquatic life and that their toxicities are additive. No data are known to be available concerning whether the toxicities of the forms of arsenic to aquatic organisms are additive. The conversion factor for arsenic (acute and chronic values), to translate the total recoverable value to the dissolved form is 1.0. The recommended exceedance frequency of 3 years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to arsenic exceeds the criterion.

For arsenic, the criteria to protect saltwater aquatic life were derived to be 36 μg/L (as chronic value), and 69 μg/L (acute value). These criteria, expressed in terms of the dissolved metal, and were adopted by the USVI into the WQSR.

(b) Cadmium

The USVI adopted cadmium (as the dissolved form of the metal) criteria of 40 μg/L (acute value) and 8.8 μg/L (chronic value) for protection of saltwater aquatic life in all marine waters (class A, B and C waters). These criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 2001 and described in EPA's Report: *2001 Update of Ambient Water Quality Criteria for Cadmium* (EPA 2001). Both criteria were adopted based on EPA's recommended 304(a) aquatic life criterion and derived using the *Guidelines* (Stephan et al. 1985).

Tests of the acute toxicity of cadmium to saltwater organisms had been conducted with 50 species of invertebrates and 11 species of fish, representing the required eight different taxonomic families. Saltwater cadmium SMAVs were available for species in 54 genera and SMAVs for 50 species of invertebrates ranged from 41.29 μg/L for a mysid to 135,000 μg/L for an oligochaete worm. SMAVs for 11 fish species ranged from 75.0 μg/L for striped bass to 50,000 μg/L for sheepshead minnow. The acute toxicity of cadmium generally increased as salinity decreased. The effect of temperature appeared to be species-specific.

Of the 54 saltwater genera for which acute values were available, the most sensitive, *Americamysis*, was 3,270 times more sensitive than the most resistant, *Monopylephorus*. The SMAVs for saltwater invertebrate species ranged from 41.29 μg/L for a mysid to 135,000 μg/L for an oligochaete worm. The acute values for saltwater polychaetes range from 200 μg/L for *C. capitata* to 14,100 μg/L for *N. arenaceodentata*. Saltwater molluscs had SMAVs from 227.9 μg/L for the Pacific oyster to 19,170 μg/L for the mud snail. Acute values were available for more than one species in each of seven genera, and the range of SMAVs within each genus was

no more than a factor of 3.6 for six of the seven genera. The seventh genus, *Crassostrea*, had two SMAVs that differed by a factor of 16.7, possibly due to different exposure conditions between species.

The saltwater FAV for total cadmium, calculated from the GMAVs was 80.55 μg/L. The resultant saltwater CMC for total cadmium was 40 μg/L. If the total cadmium CMC was converted to the dissolved form of cadmium using the 0.994 factor determined experimentally by EPA, the saltwater CMC for dissolved cadmium was 40 μg/L.

The two saltwater invertebrate mysid species for which acute-chronic ratios were available (*Americamysis bahia* and *Mysidopsis bigelowi)* had SMAVs in the same range as the saltwater FAV, and so it seemed reasonable to use the geometric mean of these two ratios. Chronic tests had been conducted with these two species, with SMCVs of 6.173 μg/L and 7.141 μg/L, respectively. ACRs were available for each species, with values of 5.384 for *A. bahia* and 15.40 for *M. bigelowi*. The saltwater FAV of 80.55 μg/L was divided by the mean ACR of 9.106, and a saltwater FCV of 8.9 μg/L was obtained. The dissolved cadmium FCV was computed using the conversion factor (0.994 x 8.846 μg/L), and was equal to 8.8 μg/L. The acute values appeared to reflect effects of varying salinity and temperature levels, whereas the few available chronic values apparently did not.

The procedures described in the *Guidelines* indicate that saltwater aquatic life should be protected if the four-day average dissolved concentration of cadmium does not exceed 8.8 μg/L more than once every three years on the average and if the 24-hour average dissolved concentration does not exceed 40 μg/L more than once every three years on the average. The recommended exceedance frequency of 3 years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to cadmium exceeds the criterion. The USVI has adopted the above criteria for dissolved concentrations of cadmium into the WQSR.

(c) Chromium (VI)

The USVI adopted chromium (VI) criteria of 1,100 μg/L (acute value) and 50 μg/L (chronic value) for protection of saltwater aquatic life in all marine waters (class A, B and C waters). These criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 1985 and described in EPA's Report: *Ambient Water Quality Criteria for Chromium -1984* (EPA 1985b).

The acute toxicity of chromium (VI) to 23 saltwater vertebrate and invertebrate species ranged from 2,000 μg/L (for a polychaece worm and a mysid) to 105,000 μg/L (for the mud Snail). Of the 21 saltwater genera for which acute values were available, the most sensitive, *Nereis*, was about 52 times more sensitive than the most resistant, *Nassarius*. This range was surprisingly small compared to the very large range of sensitivities of freshwater animals to chromium (VI). Both, the 12 most sensitive and 2 most resistant genera were invertebrates. Acute values were available for two species in each of two genera, and a range of SMAVs within each genus was

less than a factor of 2.2. The saltwater FAV of 2,158 μg/L for chromium (VI) was calculated from the GMAVs.

The life-cycle tests with the saltwater polychaece, *Neanthes arenaceodentate*, resulted in the chronic values ranging from 13 to 36.74 μg/L. The same tests with the mysid, *Mysidopsis bahia* resulted in the chronic value of 132 μg/L. The ACRs were 121.8 (for the polychaece) and 15.38 μg/L (for the mysid). These two species were among the most acutely sensitive to chromium (VI). The geometric mean of these two ratios was used as the saltwater Final ACR. The division of the saltwater FAV for chromium (VI) by the Final ACR resulted in a saltwater FCV of 49.86 μg/L.

The procedures described in the *Guidelines* indicate that saltwater aquatic organisms, and their uses should not be affected unacceptably if the 4-day average concentration of chromium (VI) does not exceed 50 μg/L (in acid-soluble form) more than once every 3 years on the average and if the 1-hour average concentration does not exceed 1,100 μg/L more than once every 3 years on the average. The recommended exceedance frequency of 3 years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to chromium (VI) exceeds the criterion. The USVI has adopted these criteria for dissolved form of chromium (VI) into the WQSR.

(d) Copper

The USVI adopted copper (as the dissolved form of the metal) criteria of 4.8 μg/L (acute value) and 3.1 μg/L chronic value for protection of saltwater aquatic life in all marine waters (class A, B and C waters).

The original EPA 304(a) aquatic life criteria for copper was published in 1985 and described in EPA's Report: *Ambient Water Quality Criteria for Copper - 1984* (EPA 1985d). These criteria were revised in 1995, based on the additional information available at that time and new, revised criteria were recommended and published in the EPA's Report: *Ambient Water Quality Criteria - Saltwater Copper Addendum*. Both sets of criteria were adopted based on EPA's recommended 304(a) aquatic life criterion and derived using the *Guidelines* (Stephan et al. 1985).

In 1985, the only saltwater chronic value for copper was available for the mysid, *Mysidopsis bahia*. The chronic toxicity of copper to this saltwater invertebrate was determined in a flow-through life-cycle test in which the concentrations of copper were measured by atomic absorption spectroscopy. Tests resulted in a chronic value of 54.09 μg/L. Using the acute value of 181 μg/L, the ACR for this species was 3.346.

Acute values for saltwater fishes ranged from 13.93 to 411.7 μg/L and, as with invertebrates, the lowest value was obtained in a test with embryos. In addict ion, tests with embryos of Atlantic cod resulted in a 14-day LC50 of 10 μg/L. The 19 available saltwater GMAVs ranged from 5.8 μg/L to 7,694 μg/L. Acute values were available for more than one species in each of five genera and the range of SMAVs within each genus was less than a factor of 3.7. A saltwater

FAV of 5.832 μg/L was obtained using the GMAVs and the calculation procedure described in the *Guidelines.* This was close to the acute value of 5.8 μg/L for the blue mussel and the value of 7.807 μg/L for the Pacific oyster.

Use of 3.346 as the saltwater Final ACR did not seem reasonable because *Mysidopsis bahia* was relatively acutely insensitive to copper. The lowest saltwater acute values were from tests with embryos and larvae of molluscs and embryos of summer flounder, which are possibly the most sensitive life stages of these species. It seemed likely that concentrations that did not cause acute lethality to these life stages of these species did not cause chronic toxicity either. Thus, for salt water the FCV for copper was equal to the CMC of 2.916 μg/L.

Based in the data obtained by EPA prior to 1985 and the procedures described in the *Guidelines,* it was the EPA's recommendation that saltwater aquatic organisms and their uses should not be affected unacceptably if the one-hour average concentration of copper does not exceed 2.9 μg/L more than once every three years on the average.

In 1995, based on new information, the above recommended values were revised by EPA. Now, there were 26 saltwater GMAVs available, an increase of six over the 1985 document. Three of the original four most sensitive genera remained so. Since measured values were now available for *Mytilus edulis*, the acute value for this sensitive species calculated in 1985 was now eliminated from SMAV. In addition, all of acute values were now adjusted to dissolved form of the metal (1984 recommendations were based on the total recoverable form of metal). There were six LC50values available now ranging from 14.9 μg/L to 21.0 μg/L, with the geometric mean of 17.7 μg/L. The GMAVs for the other sensitive species differed by factor of 2.2. Using the method of calculation outlined in the *Guidance*, the saltwater dissolved copper FAV was 10.39 μg/L. This FAV was lowered to 9.625 μg/L to protect the commercially important blue mussel. The CMC is the FAV divided by two, thus the new saltwater dissolved copper CMC was now calculated to be 4.8 μg/L.

The Final ACR was calculated to be 3.127, the geometric mean of four species ACRs. The FCV, calculated by dividing the FAV by this ratio was 3.078 μg/L. The CCC was equal to the FCV, rounded up to two significant figures, with a value of 3.1 μg/L.

The procedures described in the *Guidelines* indicate that saltwater aquatic organisms, and their uses should not be affected unacceptably if the 4-day average concentration of copper does not exceed 3.1 μg/L more than once every 3 years on the average and if the 1-hour average concentration does not exceed 4.8 μg/L more than once every 3 years on the average. The USVI has adopted these criteria for dissolved form of copper the WQSR.

(e) Lead

The USVI adopted lead criteria (as the dissolved form of the metal) of 210 μg/L (acute value) and 8.1 μg/L (chronic value) for protection of saltwater aquatic life in all marine waters (class A, B and C waters). These criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 1986 and described in EPA's "Gold Book" Report: "Quality Criteria for

Water" (EPA 1986a). The criterion for lead was derived using the "Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses" (Stephan et al. 1985).

The procedures described in, the *Guidelines* indicate that saltwater aquatic organisms and their uses should not be affected unacceptably if the 4-day average concentration of lead does not exceed 8.1 μg/L more than once every 3 years on the average and if the 1-hour average concentration does not exceed 210 μg/L more than once every three years on the average. The USVI has adopted the EPA recommended criteria for lead (in form of the dissolved metal in the water column) into the WQSR.

(f) Mercury – Methylmercury

The USVI adopted methylmercury criteria of 1.8 μg/L (acute value) and 0.94 μg/L (chronic value) for protection of aquatic life in all marine waters (class A, B and C waters). Both criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in January 1985 and described in EPA's mercury criteria document (EPA 1985c).

The saltwater CCC of 0.025 μg/L given on page 23 of the criteria document is based on the Final Residue Value procedure in the 1985 *Guidelines* (Stephan et al. 1985*).* After the conversion factor of 0.85 is applied to both, acute and chronic values, these criteria are expressed as dissolved, instead of total recoverable (which is how the original EPA criteria were expressed

The procedures described in the *Guidelines* indicate that saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of mercury (in total recoverable form) does not exceed 0.94 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed a 1.8 μg/L more than once every three years on the average. The USVI has adopted the EPA recommended criteria for mercury (in form of the dissolved metal in the water column) into the WQSR.

(g) Nickel

The USVI adopted nickel criteria (as the dissolved form of the metal) of 74 μg/L (acute value) and 8.2 μg/L chronic value for protection of aquatic life in all marine waters (class A, B and C waters). These criteria were adopted based on EPA's recommended 304(a) aquatic life criteria published in 1985 and described in EPA's Report: "Ambient Aquatic Life Water Quality Criteria for Nickel". Criteria for lead were derived using the *Guidelines* (Stephan et al. 1985).

The acute toxicity of nickel to saltwater organisms had been determined with 18 species of intervertebrates and 4 species of fish. Of the twenty saltwater genera for which acute values were available, the most sensitive genus was over 2,000 times more sensitive than the most resistant one. Acute values were available for more than one species in each of the genera, and

the range of SMAVs within each genus was less than a factor of 4.8. GMAVs for the most sensitive genera were within the factor of 7.8. The saltwater FAV was calculated to be 149.2 μg/L, which was very close to the acute value for the most sensitive tested saltwater species. Division of the saltwater FAV by two resulted in the final acute value of 74 μg/L.

The mysid, *Mysidopsis bahia* was the only saltwater species with which an acceptable chronic test had been conducted on nickel. Chronic exposure to nickel reduced survival at 141 μg/L. The chronic value for nickel with this species was 92.74 μg/L and the ACR was 5.478. The three available species mean ACRs ranged from 5.478 to 35.58 μg/L and were all determined with species sensitive to nickel. The Final ACR of 17.99 μg/L was calculated as the geometric mean of the three ratios. Division of the saltwater FAV by 17.99 μg/L resulted in the saltwater FCV of 8.293 μg/L, which was about a factor of eleven lower than the only chronic value that had been determined with a saltwater species.

The procedures described in the *Guidelines* indicate that saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of nickel (in total recoverable form) does not exceed 8.3 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed a 75 μg/L more than once every three years on the average. The USVI has adopted the EPA recommended criteria for nickel (inform of the dissolved metal in the water column) into the WQSR. The conversion factor for nickel (for both, acute and chronic values), to translate the total recoverable value to the dissolved form was 0.99.

(h) Selenium

The USVI adopted selenium criteria (as the dissolved form of the metal) of 290 μg/L (acute value) and 71 μg/L (chronic value) for protection of aquatic life in all marine waters (class A, B and C waters).

The selenium criteria document, well describing the derivation procedure for selenium criteria has been published in 1987 (EPA 1987b). In the Toxic Rule, the EPA proposed an acute criterion for selenium based on the criterion proposed for selenium in the Water Quality Guidance for the Great Lakes System (61 FR 58444), which takes into account that selenium's two most prevalent oxidation states, selenite and selenate, present different potentials for aquatic toxicity, and that that various forms of selenium are additive. The new approach produces a different selenium acute criterion concentration, or CMC, depending upon the relative proportions of selenite, selenate, and other forms of selenium that are present.

The USVI adopted the EPA recommended criteria for selenium (in form of the dissolved metal in the water column) into the WQSR. The conversion factor for selenium (acute and chronic values), to translate the total recoverable value to the dissolved form is 0.998.

(i) Silver

The USVI adopted a silver (as the dissolved form of the metal) criterion of 1.9 μg/L (acute value) for protection of saltwater aquatic life in all marine waters (class A, B and C waters). This criterion was adopted based on EPA's recommended 304(a) aquatic life criterion published in 1980 and described in EPA's Report: *Ambient Water Quality Criteria for Silver* (EPA 1980e).

For saltwater animals, acute toxicity data were available for five fish and five intervertebrate species. Fishes were both the most sensitive and the most resistant species tested, but the intervertebrate species, as a group, were generally more sensitive to silver than were the fish. Toxicity values ranged from 4 .7 μg/L for the summer flounder to 1,400 μg/L for the sheepshead minnow. The saltwater FAV for silver, derived from the SMAV, using the calculation procedures described in the Guidance, was 2.3 μg/L.

The chronic toxicity value of 18 μg/L for the saltwater mysid shrimp was determined in a flow-through, life-cycle test. Because of the variations is in the results of chronic tests with rainbow trout and the problems with determining an ACR for *Daphnia magna*, neither a Final ACR nor a freshwater or saltwater FCVs could have been determined for silver. In summary, for saltwater aquatic life, the concentration of total recoverable silver should be 2.3 μg/L. No data were available concerning the chronic toxicity of silver to sensitive aquatic life.

The recommended criterion for saltwater aquatic life states that the concentration of total recoverable silver should not exceed 2.3 μg/L at any time. Using the conversion factor of 0.85, the recommended criterion was recalculated to the concentration of the dissolved form of metal 1.9 μg/L concentration. No data were available concerning the chronic toxicity of silver to sensitive saltwater aquatic life. The USVI has adopted the EPA recommended criteria for silver (in form of the dissolved metal in the water column) into the WQSR. The conversion factor for silver (acute value), to translate the total recoverable value to the dissolved form was 0.85.

(j) Zinc

The USVI adopted zinc criteria (as the dissolved form of the metal) of 90 μg/L (acute value) and 81 μg/L (chronic value) for protection of aquatic life in all marine waters (class A, B and C waters). These criteria were adopted based on EPA's recommended 304(a) aquatic life criteria published in 1987 and described in EPA's Report: *Ambient Water Quality Criteria for Zinc-1987* (EPA 1987a). The criteria for zinc was derived using *the Guidelines* (Stephan et al. 1985).

Acceptable acute toxicity values for zinc were available for 33 species of saltwater animals including 26 species of invertebrates and *7* species of fish. The range of SMAVs for saltwater invertebrates extended from 195 μg/L for embryos of the quahog clam to 320,400 μg/L for adults of the clam *Macoma balthica*. The range of SMAVs for fish was narrower, extending from 191.4 μg/L for larvae of the cabezon to 38,000 μg/L for juvenile spot. As a general rule, early life stages of saltwater invertebrates and fish were more sensitive to zinc than juveniles and

adults. Temperature had variable and inconsistent effects on the sensitivity of saltwater invertebrates to zinc. The sensitivity of saltwater vertebrate animals to zinc decreased with increasing salinity, but the magnitude of the effect was species-specific.

A life-cycle test with the mysid, *Mvsidopsia bahia*, found unacceptable effects at 120 μg/L, but not at 231 μg/L, and the ACR was 2.997.

In summary, of the 28 genera for which saltwater GMAVs were available, the most sensitive one (*Scorpaenichthys*) was about 1,700 times more sensitive than the most resistant one, *Macoma*. Clams were both sensitive and resistant to zinc. Acute values were available for more than one species in each of five genera and the SMAVs within each genus were less than a factor of 5.2. The saltwater FAV for zinc was calculated to be 190.2 μg/L, which was slightly lower that the acute value for the most sensitive species.

The procedures described in the *Guidelines* indicate that saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of zinc (in total recoverable form) does not exceed 86 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 95 μg/L more than once every three years on the average. The recommended exceedance frequency of 3 years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to zinc exceeds the criterion.

The USVI has adopted the EPA recommended criteria for zinc (in form of the dissolved metal in the water column) into the WQSR. The conversion factor used for zinc (acute and chronic values), to translate the total recoverable values to the dissolved form was 0.946.

(2) Non-metals:

(a) Chlorine

The USVI adopted chlorine criteria of 13 μg/L (acute value) and 7.5 μg/L (chronic value) for protection of aquatic life in all marine waters (class A, B and C waters). These criteria were adopted based on EPA's recommended 304(a) aquatic life criteria published in 1986 and described in EPA's "Gold Book" Report: *Quality Criteria for Water* (EPA 1986). The criteria for chlorine were derived using *the Guidelines* (Stephan et al. 1985).

The acute sensitivities of 24 species of saltwater animals in 21 genera have been determined for chlorine, and the LC50 range from 26 μg/L for the eastern oyster to 1,418 μg/L for a mixture of two shore crab species. Twenty-one GMAVs were available for saltwater organisms. Acute values were available for more than one species in each of two genera and the range of SMAVs within each genus was less than a factor of 2.2. The most sensitive genus, *Crassostrea*, was 54 times more sensitive than the most resistant, *Hemigrapsus*. Nine of the eleven most resistant genera were invertebrates. In contrast, seven of the ten most sensitive genera were fishes. The four most sensitive genera included such economically and ecologically important species as the

coho salmon, tidewater silverside, Atlantic silverside, Acartia tonsa, and eastern oyster. These data resulted in a saltwater FAV of 25.24 μg/L, providing the final acute value of 13 μg/L

Only one chronic test has been conducted with a saltwater species, *Menidia Eeninsulae,* and in this test the ACR was 1.162. The species mean ACRs of two of the more sensitive freshwater species and the one sensitive saltwater species were all between 1.0 and 6.2. The ratio for the more resistant scud was greater than 37. Thus, it seemed reasonable to calculate the Final ACRs as the geometric mean of the three lower ratios, resulting in a value of 3.345. The resulting saltwater FCV was 7.546 μg/L.

The procedures described in the *Guidelines* indicate that except possibly where a locally important species is very sensitive, saltwater aquatic organisms and their uses should not be affected unacceptably if the four-day average concentration of chlorine-produced oxidants does not exceed 7.5 μg/L more than once every three years on the average and if the one-hour average concentration does not exceed 13 μg/L more than once every three years on the average. The recommended exceedance frequency of 3 years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to chlorine exceeds the criterion. The USVI has adopted the above criteria for chlorine into the WQSR.

(b) Ammonia

The USVI adopted ammonia criteria, which were calculated, based on total ammonia concentrations for the pH range of 7.0 to 9.0, temperature range of 0 to 35°C, and salinities of 10, 20 and 30 g/kg. Criteria were adopted based on EPA's recommended 304(a) aquatic life criterion published in 1989 and described in EPA's Report: *Ambient Water Quality Criteria for Ammonia - 1989* (EPA 1989).

The acute toxicity of ammonia to saltwater animals has been studied in crustaceans, bivalve mollusks, and fishes. Data available on the acute toxicity of ammonia to 21 saltwater animals in 18 genera showed LC50 concentrations ranging from 0.23 to 43 mg/L for winter flounder, *Pseudopleuronectes americanus*, which was the most sensitive species, with a mean LC50 (SMAV) of 0.492 mg/L. The three most tolerant species were mollusks. The SMAVs were 19.1 mg/L for the Eastern oyster, *Crassostrea virginica*, 5.36 mg/L for the quahog clam, *Mercenaria mercenaria,* and 3.08 mg/L for the brackish water clan, *Rangia cuneata*. Fishes and crustaceans were well represented among both the more sensitive and the more tolerant species tested. The mean acute sensitivity of 88 % of the species tested was within a factor of ten of that for the winter flounder. Water quality, particularly pH and temperature, but also salinity, affected the proportion of un-ionized ammonia.

In saltwater, a life-cycle toxicity test had been conducted with the mysid, *Mysidopsis bahia*, and an early life-stage test has been completed with the inland silverside, *Menidia beryllina*. The effect of ammonia on survival, growth and reproduction of *M.bahia* was assessed in a life-cycle toxicity test lasting 32 days. Survival was reduced to 35 % of that for controls and length of males and females was significantly reduced in ammonia concentration of 0.331 mg/L. Although reproduction was markedly diminished in this concentration, it did not differ

significantly from controls. No significant effects on mysids were detected at 0.092 mg/L. The chronic limits were 0.163 and 0.331 mg/L for a chronic value of 0.232. The Acute Value from a flow-through test conducted at similar coalitions (7.95 pH, 26.5°C, 30.5 g/kg salinity) with *M. bahia* was 1.70 mg/L which results in an ACR of 7.2 with this species. ACRs for the saltwater species were 7.2 for the mysid and 21.3 for inland silversides. These saltwater species have similar acute sensitivities to ammonia, with LC50s near the median for the 21 saltwater species tested.

The procedures described in the *Guidelines* indicated that, except possibly where a locally important species is very sensitive, saltwater aquatic organisms should not be affected unacceptably if the four-day average concentration of un-ionized ammonia does not exceed 0.035 mg/L more than once every three years on the average and if the one-hour average concentration does not exceed 0.233 mg/L more than once every three years on the average (Stephan et al. 1985).

(c) Cyanide

The USVI adopted a cyanide criterion of 1 µg/L, as free cyanide (acute value) and the ame concentration for chronic value for protection of saltwater aquatic life in all marine waters (class A, B and C waters).

This criterion was adopted based on EPA's recommended 304(a) aquatic life criterion published in 1986 and described in EPA's "Gold Book" Report: *Quality Criteria for Water* (EPA 1986). The criterion for cyanide was derived using the "*Guidelines for Deriving Numerical National Water Quality Criteria for the Protection of Aquatic Organisms and Their Uses*" (Stephan et al. 1985).

The acute toxicity of free cyanide to saltwater species ranged from 4.893 µg/L to >10,000 µg/L and invertebrates were both the most and least sensitive species. Long-term survival in an early life- stage test with the sheepshead minnow gave a chronic value of 36.12 µg/L. Long-term survival in a mysid life-cycle test resulted in a chronic value of 69.71 µg/L. Tests with the red macroalga, *Champia parvuIla* showed cyanide toxicity at 11 to 25 µg/L, but other species were affected at concentrations up to 3,000 µg/L.

The procedures described in the Guidelines indicate that saltwater aquatic organisms and their uses should not be affected unacceptably if the 1-hour average concentration of cyanide does not exceed 1.0 µg/L more than once every 3 years on the average. The recommended exceedance frequency of 3 years is the Agency's best scientific judgment of the average amount of time it will take an unstressed system to recover from a pollution event in which exposure to cyanide exceeds the criterion. The USVI has adopted the above criteria for free cyanide into the WQSR.

(d) Sulfide-hydrogen sulfide

The USVI adopted sulfide-hydrogen sulfide criterion of 2 µg/L chronic value for protection of aquatic life in all marine waters (class SA, SB and SC waters). Criterion was adopted based on EPA's recommended 304(a) aquatic life criterion published in 1976 and described in EPA's "Red Book" Report: *Quality Criteria for Water* (EPA1976).

On the basis of chronic tests evaluating growth and survival, the safe sulfide-hydrogen sulfide level for bluegill (*Lepomis macrochirus*) juveniles and adults was 2 µg/L. Egg deposition in blue gills was reduced after 46 days in 1.4 µg/L H2S. White sucker eggs were hatched at 15 µg/L, but juveniles showed growth reduction at 1 µg/L. Safe level for fathead minnows were between *2* and 3 µg/L. Studies showed that safe levels for *Gammarus Pseudolimnaeus* and *Hexagenia limbata* were *2* and 15 µg/L, respectively. Some species typical of normally stressed habitats, *Asellus spp.*, were much more resistant.

It was recognized that the hazard from hydrogen sulfide to aquatic life was often localized and transient. Available data indicated that water containing concentrations of 2.0 µg/L undissociated H2S would not be hazardous to most fish and other aquatic wildlife, but concentrations in excess of 2.0 µg/L would constitute a long-term hazard. The USVI has adopted the above recommended criteria for sulfide-hydrogen sulfide into the WQSR.

IX. Assessing effects of the USVI Water Quality Standards on Federally-listed species

A. Narrative Water Quality Standards:

1. Water Quality Criteria

In WQSR section 186-5 (a) the USVI adopted narrative water quality criteria, which are applicable to all Territorial waters to ensure that they will meet generally accepted aesthetic qualifications and will be capable of supporting diversified aquatic life. According to section 186-5(a)(1), all waters of the USVI should be free of the following substances:

- Deposits - materials that will settle to form objectionable deposits,
- Matter - floating debris, oils, scum, and other nuisance matter,
- Turbidity - substances producing objectionable turbidity,
- Materials - including radionuclides, in concentrations or combinations which are toxic or which produce undesirable physiological responses in human, fish and other animal life, and plants,

- Color - virtually free from substances producing objectionable color for aesthetic purposes,
- Suspended, colloidal, or settleable solids - from wastewater sources which will cause disposition or be deleterious for the designated uses,
- Oil and floating substances - residue attributable to wastewater or visible oil film or globules of grease,
- Taste and odor producing substances - in amounts that will interfere with the use for primary contact recreation, potable water supply or will render any undesirable taste or odor to edible aquatic life,
- Substances and/or conditions - in concentrations which produce undesirable aquatic life, and
- Nuisance species - Exotic or aquatic.

2. Biological Criteria (biocriteria)

In addition, the USVI also adopted narrative biocriteria into their WQSR, section 186-5 (b). These narrative biological criteria apply to all marine and coastal waters of the USVI, including estuarine, mangrove, seagrass, coral reef and other marine ecosystems based upon their respective reference conditions and metrics. According to this criterion, all of marine waters of the USVI shall be of a sufficient quality to support a resident biological community as defined by metrics based upon reference conditions. The condition of these waterbodies shall be determined from measures of physical, chemical, and biological characteristics of each waterbody class, according to its designated use.

The biological integrity of the benthic communities living within waters may be considered as a component of these measures. These communities shall be assessed by comparison to reference conditions(s) with similar abiotic and biotic environmental settings that represent the optimal or least disturbed condition for that system. Such reference conditions shall be those observed to support the greatest community diversity, and abundance of aquatic life as is expected to be or has been historically found in natural settings essentially undisturbed or minimally disturbed by human impacts, development, or discharges. This condition shall be determined by consistent sampling and reliable measures of selected indicator communities of flora and/or fauna and may be used in conjunction with other measures of water quality.

For Class A waters, the primary goal is to preserve the unique characteristics of the waters designated as outstanding Natural Resource Waters (e.g., Natural Barrier Reef at Buck Island, St. Croix and the Under Water Trail at Trunk Bay, St. John), waters of exceptional recreational, environmental or ecological significance.

For Class B waters, minimal changes in structure of the biotic community and minimal changes in ecosystem function are allowed. Virtually, all native taxa are maintained with some changes in biomass and/or abundance; ecosystem functions are fully maintained within the range of natural variability.

For Class C waters, evident changes in structure of the biotic community and minimal changes in ecosystem function are allowed. Evident changes in structure due to loss of some rare native taxa; shifts in relative abundance of taxa (community structure) are allowed but sensitive-ubiquitous taxa are expected to remain common and abundant; ecosystem functions are expected to be fully maintained through redundant attributes of the system.

In summary, the narrative water quality criteria and biological criteria identified above and published in the VIWQSR section186-5, describe the goals including the physical, chemical and biological conditions and quality that all waters of the USVI must attain. Narrative criteria define the desired aesthetic qualities and properties of all waters and provide basis for prohibiting undesirable conditions, where numeric criteria may not be available or where the cause of the nuisance or toxicity is not specifically known. The CWA goals are to achieve the highest chemical, physical and biological quality for state waters and to achieve a level of water quality that provides for the protection and propagation of fish, shellfish and wildlife and recreation in and on the water. These provisions fulfill the regulatory directive to establish supplemental narrative criteria in addition to numeric criteria. These provisions help to protect aquatic life, wildlife, water quality as well as human health.

Conclusion: EPA determined that the listed above narrative criteria, adopted by the USVI, offer additional protection for Nassau grouper and sixteen ESA-listed species which are subject to this consultation, as well as their habitats, especially in cases where the numeric criteria are not yet derived or adopted. As a result, the EPA determined these criteria to be beneficial to the VI's marine environment and not likely to adversely affect (NLAA) listed species or their critical habitats. These narrative criteria can be numerically interpreted to provide protection to the aquatic communities of the USVI. EPA is currently working closely with the VIDPNR to identify the appropriate indicators and derive the protective thresholds to allow for the adoption of the numeric biological criteria in the future.

B. Numeric Water Quality Standards:

For the purpose of this consultation, the EPA used all best available scientific and commercial data to determine whether or not the numeric water quality criteria adopted by the USVI into the WQSR (2015) are protective of the seventeen marine species of interest. Based on the scientific literature review, the EPA attempted to establish the optimal criteria desired for the recovery of Nassau grouper and ESA-listed species and compared them to the applicable criteria adopted by the USVI to determine whether or not they are fully protective of the aquatic life designated use. In some cases, this process has been proven to be challenging due to the great uncertainty associated with projected changes in marine water quality due to climate change, as well as great uncertainty associated with specific threats and, in many cases, very limited species-specific information on responses to these threats. In cases where there was not enough information available to make a determination, the EPA clearly stated this and relied on the best professional judgment to extrapolate available data to the species of interest.

1. Dissolved Oxygen

The amount of dissolved gases in marine waters varies according to the types of life forms present in the seawater (plants and animals) and their relative proportions. Most marine species need dissolved oxygen (DO) to survive, and continually consume it from the waterbodies that they live in. Replenishment of DO comes from the daytime photosynthetic activity of plants and from surface diffusion. If there is a large number of plants present in the given waterbody, then the daytime DO levels can be high. On the other hand, if there are few plants but a large number of animals present in the waterbody then DO levels can be low. Overall, the DO concentrations in marine waters can range from 0 milligrams per liter (mg/L) to over 20 mg/L(Haas et al. 2010). The maximum amount of DO present in the water, increases, as the water temperature decreases, resulting in cold waters being able to hold more oxygen than warmer waters. This may potentially become an issue with the upcoming climate changes and warming up of the marine waters. Long-term effects of these changes on marine species are however unknown.

a) Analysis of potential impacts of DO on Corals

All seven ESA-listed coral species are hermatypic (reef building) corals. These types of coral polyps create mutualistic relationship with zooxanthellae living within them (Muller-Parker and D'Elia 1997). These zooxanthellae corals are fully dependent on the photosynthetic algae for their respiration. Coral polyps produce carbon dioxide (CO2) and water as byproducts of cellular respiration. The zooxanthellae cells use both byproducts to carry out photosynthesis, resulting in the production of sugars, lipids and oxygen. These, on the other hand, are being used by coral polyps to respire and grow. The tight recycling of all of the above products between the coral polyp cells and the zooxanthellae is the driving force behind the growth and productivity of coral reefs.

According to the literature, due to day time photosynthesis production and night time plant respiration, DO levels surrounding coral reefs can fluctuate from 4-15 mg/L, but usually remain around 5-8 mg/L (Haas et al. 2010). Due to high photosynthetic activity and aeration from breaking waves in shallow marine environments, DO concentrations found close to coral reefs are generally higher when compared to DO levels measured in the surrounding water areas (Haas et al. 2014). It has been reported in the literature, however, that DO concentrations at close proximity to coral structures may, in some cases, reach concentrations well below saturation level (Haas et al. 2010). In complex coral reef environments, in situ oxygen levels can fluctuate widely throughout the daily cycle and can be very low at night when respiration dominates. The extent of this diurnal fluctuation is largely influenced by the biological nature of the local benthic community. For example, DO concentrations in water surrounding algae-dominated areas can be lower than DO concentrations measured in adjacent coral-dominated areas.

These observations support results of the research published by Barott et al. (2009). The DO levels reported in this publication (measured at the coral/algae interface) were clearly dependent on the interactions between these two species. Barott et al. reported that in cases when the algae

were the superior competitor, average DO concentrations measured at coral/algae interface were 3.2±0.5 mg/L and 2.9±0.4 mg/L. However, when both organisms (algae and corals) were in a stable state or when corals were the superior competitor, DO concentrations measured at their interfaces were on average 7.9±0.7 mg/L, well above the suggested threshold of 4 mg/L rates (Barott et al. 2009; Haas 2010).

It is important to point out that, in addition to the diurnal variability in DO levels described above, sources of low DO observed in the coral reef areas are often anthropogenic. These low-DO zones can be the result of a fertilizer-fueled algae and high phytoplankton growth. When the algae and phytoplankton die, they are being decomposed by microbes at the seafloor using up the oxygen. Although there is not much information published in the scientific literature, which would describe the individual responses of corals and algae to the reduced DO conditions, there is a strong evidence indicating that hypoxia plays an important role in coral-algae competition, impacting structures of coral reef communities (McCook 2001). The mechanism of interactions between these two species was well described by Falkowski et al. (1984). Based on this research, under low DO conditions, algal metabolic by-products fuel metabolism of surrounding microbial community, potentially leading to high bacterial concentrations around the coral structures, thus increasing chances for coral infections and massive die-offs.

Based on the information published in the scientific literature, the conclusion can be made that corals, in general, are not able to survive when levels of DO drop below 2 to 5 mg/L for a longer periods of time (Haas 2010; Recyclers 2011; Ichthyology 2011). When this occurs, aquatic organisms die in large numbers, decreasing community diversity further affecting established food chains. Once the herbivorous fish that kept harmful algae in check are eliminated, algae will overgrow and smother the coral reefs (Recyclers 2011). Overall, research results indicate that corals could tolerate reduced oxygen concentrations, but only until a given threshold determined by a combination of exposure time and DO concentration. Exceeding this threshold (which is reported to be 4 mg/L in multiple studies) led to rapid loss of coral tissue and higher mortality rates (Barott et al. 2009; Haas 2010; Recyclers 2011).

Haas et al. investigated the reactions of coral *Acropora yongei* and the green alga *Bryopsis pennata* species to different levels of oxygen, experimentally manipulated to imitate oxygen conditions reported to occur in coral reef environments (Haas et al. 2014). In this study, the hypoxic condition was defined with DO concentrations below 2 mg/L, which was based on DO concentrations observed around coral reef ecosystems at night, at the location of coral/algal interaction zones. Results showed that very low DO concentrations (ranging from 2 to 4 mg/L) had severe impacts on coral specimens over a short period of time, however, researchers reported that corals were able to tolerate DO concentrations, ranging from 4 to 6 mg/L, "reasonably well". These night time oxygen concentrations, which were commonly found during the early morning hours in various reef locations, showed no effects on the physiological performance of coral specimens over the experimental period over ten consecutive diurnal cycles.

In summary, as a result of the process of the scientific literature review, the EPA did not identify any scientific studies which focused on the identification of the optimal DO conditions for the recovery of any of the seven ESA-listed coral species of concern. The most relevant research was one performed on *Acropora yongei* specimens. The genus Acropora represents over 149 stony coral species, which include two of the ESA-listed species of concern: *Acropora palmata*

(Elkhorn coral) and *Acropora cervicornis* (Staghorn coral). Coral species grouped in Acropora genus share many similar characteristics. They all are colonies of individual polyps building the calcium carbonate sub-structures, are most common in shallow reef environments with bright light and moderate to high water motion (Richards et al. 2008). As a result, for the purpose of this consultation, the EPA considered both *A. palmata* (Elkhorn coral) and *A. cervicornis* (Staghorn coral) species to be as sensitive to the DO conditions as *A. yongei* species.

In addition, the EPA was not able to locate any publications directly related to the optimal DO conditions for the recovery of the remaining five, newly listed, coral species of concern: Pillar coral, *Dendrogyra cylindrus;* Lobed star coral, *Orbicella annularis;* Mountainous star coral, *Orbicella faveolata*; Boulder star coral, *Orbicella franksi* and Rough cactus coral, *Mycetophyllia ferox*. Taking into consideration that the above five coral species, according to NOAA-NMFS, are exposed to and affected by the same environmental threats as threats identified for Acropora species, and after detailed evaluation of anatomic and habitat characteristics for all three genus: Dendrogyra, Orbicella and Mycetophyllia, the EPA considered the information relative to Acropora species to be relative to all ESA-listed coral species of interest. There was no information identified during the scientific literature review, precluding the Agency from assuming that these five coral species will be protected by the DO criterion adopted by the USVI to the same degree as the Acropora coral species.

Conclusion: Although multiple stressors have been identified by NOAA-NMFS, in the listing process, as environmental threats to ESA-listed coral species, the oxygen levels present in the marine ambient waters were not identified as a direct threat to the recovery of any of the ESA-listed species. The threshold of 4 mg/L was reported as protective by multiple researchers, however, the optimal DO concentrations desired for the recovery of the individual coral species were not identified by any of the researchers. Overall, the EPA did not find in the literature any indication which would suggest that DO concentrations at 5.5 mg/L or 5.0 mg/L was ever identified as a threat to the recovery of the ESA-listed coral species.

In summary, based on the scientific information summarized above, the EPA determined that the numeric DO criterion of not less than 5.5 mg/L, adopted by the USVI for Class A and B waters, and the DO criterion of not less than 5 mg/L, adopted by the USVI for Class C waters, are beneficial to marine environment and result in full protection of seven ESA-listed coral species and their habitats. As a result, the EPA considers the USVI numeric DO provisions NLAA ESA-listed coral species or their critical habitats.

b) Analysis of potential impacts of DO on Whales

Whales are air-breathing marine mammals, which need to surface in order to breathe. As a result, their respiration is not affected by the DO concentrations present in the ambient water. Indirect impacts of the DO concentrations present in the waters are expected to be related to changes in whale's habitats and shifts in the spatial distribution of their prey (for both, baleen and tooth whales). With changing climate and warming water temperatures, DO levels in seawater are expected to decrease, making enormous stretches of deep ocean more hostile to whales prey.

The specific effects of changes in DO on whales' habitat and distribution of their prey is yet to be examined.

In general, whales are highly migratory and as such can easily relocate to the most optimal water conditions. In addition, whales have tendency to dive into deep waters, thus their bodies are adjusted to sudden DO changes occurring in deeper, colder waters.

- Sperm whale, *Physeter macrocephalus*

Sperm whale is the only species out of five ESA-listed whales which is a toothed whale. As a result, it feeds on large prey such as large squid and fish, including some species of sharks.

As stated by NMFS (2010b) in the Final Recovery Plan for the Sperm Whale, the impact of climate change on sperm whale prey "continues to be studied". Due to increasing low-oxygen zones in deep ocean, researchers expect prey to "relocate" closer to the ocean surface. Because the distribution range of sperm whales is extensive, this species is expected to be more resilient to climate change (thus, decreased DO) than a species with a narrower distribution range. In general, the optimal DO condition for whales is the one which is optimal for their prey. Giant squid comprise about 80% of the sperm whale diet and has a high oxygen demand (Guerra et al 2011). The remaining 20% of sperm whale diet is comprised of octopus, fish, shrimp, crab and even small bottom-living sharks (http://www.afsc.noaa.gov/nmml/education/cetaceans/sperm.php.). The DO requirements of this open-ocean and deep-ocean prey greatly vary. The reported optimal DO for shrimp is 3-4 mg/L (Chieng 1992). Sharks were reported to be limited in dive depths due to DO levels above 1.5 mg/L (Nasby-Lucas et al. 2009), which DO requirements for fish varied greatly. For example, Billfish swim in areas with a minimum of 3.5 mg/L DO, while marlins and sailfish will dive to depths with DO concentrations of 1.5 mg/L (Courtney and Brodziak, 2010). Albacore tuna live in mid-ocean levels, and require a minimum of 2.5 mg/L (Fisheries and Aquaculture Department 2000). Halibut can maintain a minimum DO tolerance threshold of 1 mg/L (Sadorus 2012).

- Blue whale, *Balaenoptera musculus;* Fin whale, *Balaenoptera physalus;* Sei whale, *Balaenoptera borealis* and Humpback whale, *Megaptera novaeangliae*

Unlike sperm whale, the remaining four ESA-listed whale species belong to baleen group of filter-feeders. Blue whales eat mostly krill. Fin whales eat krill, copepods, squids, and variety of small schooling fishes. Humpback whales prey mostly on krill and small schooling fishes. Sei whales eat copepods, krill and amphipods (another type of small crustacean). As smaller organisms, with lower oxygen needs and higher surface area-to-volume ratios, krill is less sensitive to the low-oxygen waters. As a result, baleen whales would appear to be as sensitive or less sensitive to low DO levels when compared to the toothed sperm whale.

Conclusion: There is no information in the scientific literature that would suggest that the DO criteria adopted by the USVI are not protective and potentially pose a threat to the survival of the

five ESA-listed whale species. Taking into consideration all of the above information, EPA has determined that the numeric DO criterion of no less than 5.5 mg/L adopted for class A and B waters and DO criterion of no less than 5.0 mg/L adopted for class C waters, are beneficial to the marine environment and result in full protection of whales and their habitats. As a result, EPA considers the U.S.V.I numeric DO provisions NLAA Sperm, Blue, Fin, Sei and Humpback whales or their critical habitats.

c) Analysis of potential impacts of DO on Sea Turtles

Sea turtles are air-breathing reptiles, which need to surface to breathe. As a result, their respiration is not directly affected by the changes in ambient DO concentrations in the surrounding water. There are no optimal DO conditions identified for the ESA-listed sea turtle species in the scientific literature, however, research suggests that sea turtle species may be impacted by low DO concentrations indirectly by loss or degradation of their habitat and resulting shifts in the distribution of their prey. As reported by researchers, coral reef ecosystems are expected to be most sensitive to the environmental changes. This sensitivity of corals to the changes in water quality is especially of concern for two of our ESA-listed sea turtle species, which are either frequent or regular visitors to the coral reef ecosystems: Hawksbill turtle (*Eretmochelys imbricata*), which feeds in the lagoon or back reef zone of coral reef ecosystems and the green sea turtle (*Chelonia mydas*), which feeds primarily on the seagrasses found in protected back reef lagoons. The impacts of ambient concentrations of DO on sensitive coral reef ecosystems were briefly described in the earlier section of this document. The DO criteria adopted by the USVI were determined to be protective of coral reef ecosystems.

In addition to the degradation of their habitat, sea turtles can also be indirectly impacted by the low levels of DO via shifts in the spatial distribution of their prey. Craig et al. documented the spatial distribution of bottom DO concentrations in the Gulf of Mexico and related this to the spatial distribution of demersal fish, invertebrates, cetaceans and sea turtles including the three ESA-listed species of interest: loggerhead turtle, *Caretta caretta*; leatherback turtle, *Dermochelys coriacea* and green turtle, *Chelonia mydas* (Craig at al. 2001). Hypoxia (defined as DO concentrations less than 2.0 mg/L) was widespread during June and July of 1992 and 1993 in the northwestern Gulf of Mexico, extending over 8 to 9% of the continental shelf. Analysis of data suggested that demersal species were displaced from hypoxic bottom waters to adjacent areas with intermediate levels of DO (ranging from 2.0 to 5.0 mg/L) during June-July, but returned to the original areas by October and November when hypoxia was largely absent. Aerial survey sightings suggested that chronic, large-scale hypoxia should be included among hypotheses to explain the distribution patterns of sea turtles in the northwestern Gulf of Mexico.

A substantial number of sea turtles (including Kemp's ridley, loggerhead, leatherback and green turtles) were sighted during aerial surveys in September and October, however none were observed in areas that were hypoxic the prior June-July. Turtle sightings were dominated by Kemp's ridley and loggerhead turtles, which feed in the benthos, primarily on crabs and could have been impacted by low DO concentrations via effects on the distribution of benthic food resources.

The most significant (indirect) effect of DO on sea turtles is the availability and distribution of their prey. Hawksbill turtles are mainly found on and around coral reefs feeding primarily on sponges. Green turtles are herbivores as adults forage among seagrass beds and nearshore habitats feeding primarily on algae, seagrasses, and seaweed. Hatchlings are omnivores, eating both plant and animal material. Leatherbacks, which are deep divers, are found in pelagic (open ocean) environments where they feed exclusively on jellyfish and other soft-bodied invertebrates that float in the water column (tunicates and sea squirts). Loggerhead turtles are carnivores (adults), eating crabs, conchs, whelks, and horseshoe crabs. Hatchlings are omnivores, eating both plant and animal material. There are no reports in the literature that would indicate DO levels adopted by the USVI to be a threat for any of the prey identified above.

Conclusion: There is no information in the scientific literature suggesting that the DO criteria adopted by the USVI are not protective and potentially pose a threat to the survival ESA-listed sea turtle species. Taking into consideration all of the above information, EPA has determined that the numeric DO criterion of no less than 5.5 mg/L adopted for class A and B waters and DO criterion of no less than 5.0 mg/L adopted for class C waters, are beneficial to marine environment and result in full protection of sea turtles and their habitats. As a result, EPA considers the U.S.V.I numeric DO provisions NLAA ESA-listed sea turtles or their critical habitats.

d) Analysis of potential impacts of DO on Fish

- Scalloped Hammerhead Shark, *Sphyrna lewini*

As with other fish, the scalloped hammerhead sharks obtain DO from water through gills. As they swim forward, water is forced through the gills, which extracts oxygen from the water, enabling them to breathe. As a result, they are fully dependent upon DO present in the water for their respiration. There are no optimal DO conditions identified in the scientific literature for this fish species. Research indicates, however, that due to their high mobility and body adjusted to deep dives and sudden temperature and DO changes, this species respiration ability is not likely to be affected by variability in DO levels in the ambient waters due to climate change.

Jorgensen et al. (2009), have investigated the deep water travel of the scalloped hammerhead shark to the hypoxic zone in the Gulf of California. This study demonstrated that the scalloped hammerhead shark can inhabit a highly expanded, vertical area in the open ocean, tolerating large fluctuations in depth, temperature and extremely low levels of DO. Prior to these observations, the scalloped hammerhead sharks were known to visit depths of up to 475 m, but the amount of time sharks spent in such a condition was unknown. During the study, researchers had a chance to observe the similar dives repeated over the 74-day tracking period, which suggests that low DO environment (including hypoxic zones) may be an important habitat for this species of shark, potentially to pursue prey such as deep-water squids, with no competition from other predators.

The second possible mechanism of scalloped hammerhead sharks to be affected by DO conditions discussed in the scientific literature was the expected shift in the distribution of their prey. Although such shifts are observed, they are not expected to impact the survival of scalloped hammerhead sharks. This is because this species is highly mobile, dives up to 900 m depths and relies on multiple sources of prey, making it easy to relocate towards most optimal conditions, at any given time. In the Petition to List the scalloped hammerhead shark under ESA, the authors reported that this species takes a wide variety of fish and invertebrates prey including: conger eels, milkfish, sea catfish, silversides, halfbeaks, mullet, lizardfish, barracuda, bluefish, Spanish mackerel, jacks, porgies, mojarras, cardinal fishes, goatfish, grunts, damselfishes, parrotfishes, wrasses, butterfly fishes, surgeonfish, gobies, flatfish, sharpnose sharks, blacktip reef sharks, angel sharks, stingrays, squid, octopi, cuttlefishes, sea snails, shrimp, mantis shrimp, crabs, lobsters, and isopods (Wild Earth Guardians and the Friends of the Animals 2011). As a result, even if some of the listed above prey becomes affected by DO, based on the high range of its mobility, the possibility of scalloped hammerhead shark being affected by low DO conditions are highly unlikely.

- Nassau Grouper, *Epinephelus striatus*

Similar to Scalloped Hammerhead shark, Nassau grouper also obtains DO from water through gills. This species is highly dependent on DO concentrations in ambient water for its respiration. In addition, Nassau groupers are an essential part of the coral reef ecosystem, while their juveniles depend on near-shore seagrass beds for suitable nursery habitat. The potential degradation of the Nassau grouper habitat due to low DO concentrations present in the ambient water is a significant threat for this species.

As with many other tropical saltwater fish, Nassau grouper requires higher levels of DO, such as those surrounding coral reefs (NOAA-CoRIS 2012). Coral reefs are found in the euphotic zone, where higher concentrations of DO are generally found due to high photosynthetic activities and aeration from eddies and breaking waves. As it was already described in earlier sections of this document, DO levels observed nearby coral reefs can significantly fluctuate (ranging from 4 to 15 mg/L), cycling between day photosynthesis production and night plant respiration. The optimal DO conditions reported for Nassau grouper in the literature are ranging from 4 to 8 mg/L (Long et al 2013). The DO criterion adopted by the USVI was determined to be protective of sensitive coral reef ecosystems.

Conclusion: There is no information in the scientific literature suggesting that the DO criteria adopted by the USVI are not protective and potentially pose a threat to the survival of Scalloped Hammerhead shark or Nassau grouper species. Taking into consideration all of the above information, EPA has determined that the numeric DO criterion of no less than 5.5 mg/L adopted for class A and B waters and DO criterion of no less than 5.0 mg/L adopted for class C waters, are beneficial to marine environment and result in full protection of Scalloped Hammerhead shark and Nassau groupers and their habitats. As a result, EPA considers the U.S.V.I. numeric DO provisions NLAA ESA-listed Scalloped Hammerhead shark and Nassau groupers or their critical habitats.

2. pH

The continuous increase of carbon dioxide CO2 concentrations in the atmosphere causes the increased concentrations of carbonic acid in the ocean, reducing the pH and increasing the acidity of seawater, resulting in the ocean acidification (Kleypas *et al.* 2006). An increase in the concentration of CO_2 in the water leads to an increase in the concentration of two chemicals: bicarbonate and hydrogen ions. By increasing the concentration of hydrogen ions, the pH of seawater is lowered (thus it becomes more acidic). This shift in equilibrium towards bicarbonate and hydrogen ions also causes a shift in the chemistry of calcium and carbonate ions. Hydrogen ions react with available carbonate ions to produce more bicarbonate, a process which reduces the formation of solid calcium carbonate (Marubini and Atkinson 1999).

According to the SCOR (2009) the ocean pH level of 8.179 reported back in 1751 has decreased to the pH level of 8.069, resulting in 30 % increase in acidity. Multiple publications report that the present surface seawater pH is 0.1 units lower than pre-industrial values and is predicted to further decrease by up to 0.4 units by the end of the century (IPCC 2007). Mean surface pH in the tropics (20°N to 20°S) is projected to decline from the current pH of approximately 8.05 to approximately 7.95 by 2050, and to approximately 7.75 by 2100, or a reduction of 0.31 (statistical range of 0.30 to 0.32) by 2100 (IPCC, 2013).

Researchers expect that this change in pH will likely result in changes in the physiology of ocean organisms, in particular, organisms that build their skeletons/shells from calcium carbonate, such as corals. In general, ocean acidification is reported to have potentially significant impact on the marine environment, over time.

a) Analysis of potential impacts of pH on Corals

In general, the impact of ocean acidification resulting from rising atmospheric CO2 represents a serious impediment to the recovery of ESA-listed coral species (NMFS 2014). Climate-related ocean acidification has been identified by NMFS as one of the major environmental threats for these stony corals. Although researchers around the world are aware of this threat and a lot of research is already being done to try to understand the mechanisms and the long-term impacts of pH changes in marine ecosystems, there are still many gaps in the information, which will need to be filled in order to better understand how this threat will impact the recovery of the individual coral species and the reef ecosystem, as a whole.

- Elkhorn coral, *Acropora palmata*

In the Final Listing Rule (2014), NMFS indicated that the ocean acidification will likely impact fertilization, settlement success, and post-settlement growth of *A. palmata*, thus this species was determined to be highly susceptible to acidification.

The magniture of CO2 disequilibrium between the atmosphere and ocean water is often expressed by the difference in partial pressure of the pCO2 in the ocean water (expressed in parts per million, ppm) and the partial pressure of pCO2 in overlying air (expressed in microatmospheres, μatm). This difference represents the termodynamic driving potential for CO2 gastransfer across the sea surface. The pCO2 and the pH of the seawater have the negative relationship.

Albright et al. (2010) reported that increased CO2 substantially impaired fertilization and settlement success in *Acropora palmata*. The objective of this research was to investigate the effect of CO2 on fertilization, settlement, and post-settlement growth of Caribbean elkhorn coral, at three different CO2 levels. This study demonstrated that that ocean acidification will likely impact: larval availability (by compromising fertilization); settlement ecology (by reducing settlement success) and post-settlement ecology (by impeding post-settlement growth).

Albright et al. compared to controls at pCO2 of 400 μatm (representing pH of 8.1), study reported 7% decrease in the fertilization success at the mid-CO2 level [~560 μatm] and 12% at the high-CO2 [~800 μatm] level. Mid- and high- treatments reduced the rate of fertilization and settlement (combined 52% and 73%, respectively) and post-settlement growth (39% and 50%, respectively) of *A. palmata* in lab experiments, and impairment of fertilization was exacerbated at lower sperm concentrations. Ocean acidification was shown to decrease settlement success by 45 to 69% at CO2 concentrations expected for the middle and the end of this century. The researchers further translated these compounding effects on early life history stages into a 52 to 73% reduction in the number of larval settlers on the reef. Because three-quarters of reef-building coral species spawn gametes and rely on external fertilization and planktonic development, Albright et al. considered elkhorn coral to be a representative of spawning species in general, and thus reported that the fertilization and recruitment success of many spawning corals will likely be impaired as CO2-driven ocean acidification intensifies over time.

Medina-Rosas et al. (2013) evaluated the effects of decreased pH (caused by decreased CO2) on the development of newly fertilized eggs of *A. palmata*. Three levels of CO2 enrichment were used in this study: present day conditions (400 μatm, pH 8.1) and two CO2-enriched conditions (700 μatm, pH 7.9, and 1000 μatm, pH 7.7). No effects on the progression or timing of development, or embryo and larval size, were detected in any of the three experimental runs. The results show that the embryos and larvae of *A. palmata* are able to develop normally under seawater pH of at least 0.4 pH units lower than the present levels. *A. palmata* larvae do not usually begin to calcify after settlement, so this study only examined the non-calcifying part of the life cycle of this species. Most of the concerns about the effects of ocean acidification on marine organisms is related to decreased calcification rates, thus the negative effects of decreased pH on the embryonic development of this species were not found and they may not manifest until the newly settled polyps begin to calcify.

- Staghorn coral, *Acropora cervicornis*

In the Final Listing Rule (2014), NMFS indicated that *A. cervicornis* is susceptible to ocean acidification through reduced growth, calcification, and skeletal density. NMFS concluded that *A. cervicornis* is highly susceptible to acidification.

Renegar and Riegl (2005) performed laboratory experiments to examine the effect of nutrients and CO2 on growth of *A. cervicornis*, maintained in the laboratory. Researchers measured coral growth before, during, and after exposure to elevated nitrate (5 and 10 micromolar [µM]), phosphate (2 and 4 µM) and/or CO2 (approximately 700 to 800 µatm). Researchers reported significantly reduced growth under CO2 levels of 700 to 800 µatm, predicted to occur this century, compared to controls. In addition, when elevated CO2 was combined with increased nitrate and phosphate, growth rates were further reduced. The effect of combined nitrate, phosphate, and CO2 appeared to be antagonistic, at lower nutrient concentrations and additive, at higher concentrations (compared to those nutrients paired with CO2 separately). Growth rate recovery was greater after exposure to increased nutrients or CO2 compared to increased nutrients and CO2. All corals in the combined nitrate, phosphate, and CO2 treatment experienced total mortality, indicating the severe stress this combination induced.

The effects of light and elevated CO2 on the growth and photochemical efficiency of *A. cervicornis*, were examined by Enochs et al. (2014). Corals were subjected to high and low treatments of CO2 and light. Calcification rates, linear extension, as well as colony surface area and volume of *A. cervicornis* were highly dependent on light intensity. At CO2 levels projected to occur by the end of the century from ocean acidification, *A. cervicornis* exhibited depressed calcification, but no change in linear extension. In addition, high CO2 caused depressed skeletal density, but not linear extension, illustrating that the measurement of extension by itself is inadequate to detect CO2 impacts. The skeletal integrity of *A. cervicornis* was found to be impaired, which may further reduce the resilience of the already diminished populations of this endangered species.

- Pillar coral, *Dendrogyra cylindrus*

The EPA was not able to locate any species-specific information in the scientific literature relevant to the susceptibility of *D. cylindrus*, to ocean acidification (thus, to changes in the pH levels) and its effects on the recovery of this species. No indication was found in the literature suggesting that this species is more susceptible to ocean acidification when compared to other ESA-listed corals of concern. In addition, *D. cylindrus* species is the only species within Dendrogyra genus, thus the EPA had no ability to compare a sensitivity to pH levels of this species to others species within Genus. For the purpose of this consultation, to ensure full protection, the EPA determined *D. cylindrus* to be as susceptible to ocean acidification as the remaining six ESA-listed species.

- Lobed star coral, *Orbicella annularis*

In the Final listing rule (2014), NMFS concluded that *O. annularis* is likely to have a high susceptibility to ocean acidification.

The EPA was not able to locate any species-specific information on the susceptibility of *O. annularis*, to ocean acidification (thus, to changes in the pH levels) and its effects on the recovery of this species. However, the information obtained from the literature for the other species within Orbicella genus suggest that this species is, in general susceptible to ocean acidification both through reduced fertilization of gametes and reduced growth of colonies.

- Mountainous star coral, *Orbicella faveolata*

In the Final listing Rule (2014), NMFS concluded that *O. faveolata* is highly susceptible to ocean acidification.

In laboratory experiments, reproduction of *O.faveolata* was negatively impacted by increasing CO2 and the impairment of fertilization was exacerbated at lower sperm concentrations (Albright 2011). Fertilization success was reduced by 25% at 529 μatm (with 43% fertilization) and 40% at 712 μatm (with 34% fertilization) compared to controls at 435 μatm (with 57% fertilization).

In 2011, Albright published a synthesis of the primary literature reporting on the effects of ocean acidification on sexual reproduction and early stages of corals, including work done with specimens of *O.faveolata*. Although the overall focus (and level of scientific detail) of this publication reaches outside of the scope expected for this consultation, this publication provides a basic understanding of how coral recruitment may respond to ocean acidification, in general. For areas that are deficient in studies and for which a robust assessment is not possible, lessons learned from studies conducted on more thoroughly studied taxa (e.g., mollusks, echinoderms) are used to supplement the coral literature and to provide guidelines for future experiments.

- Boulder star coral, *Orbicella franksi*

The Final listing rule (2014), NMFS concluded that there is uncertainty about how *O. franksi* will respond to ocean acidification. Based on the negative effects of acidification on growth of most corals, *O. franksi* likely has some susceptibility to acidification, however the available information did not support a more precise description of susceptibility to this threat.

The EPA was not able to locate any species-specific information on the susceptibility of *O. franksi*, to ocean acidification (thus, to changes in the pH levels) and its effects on the recovery of this species. However, the information obtained from the literature for the other species within Orbicella genus suggest that this species is, in general susceptible to ocean acidification both through reduced fertilization of gametes and reduced growth rate of colonies.

- Rough cactus coral, *Mycetophyllia ferox*

In the Final listing Rule (2014), NMFS concluded that here is uncertainty about how *M. ferox* will respond to ocean acidification. Based on the negative effects of acidification on growth of most corals, *M. ferox* likely has some susceptibility to acidification, however, the available information did not support a more precise description of susceptibility to this threat.

As indicated by NMFS, no specific research has addressed the effects of acidification on the genus Mycetophyllia. However, most corals studied have shown negative relationships between acidification and growth, and acidification is likely to contribute to reef destruction in the future. While ocean acidification has not been demonstrated to have caused appreciable declines in coral populations to date, it is considered to become a significant threat to corals by 2100.

Due to the fact that the species-specific literature presented above is very limited, the EPA reviewed the additional publications which further helped the Agency to understand the impacts of the ocean acidification on overall recovery of the reef-building corals. There are numerous mechanisms presented in the literature describing the effects of these chemical changes taking place in the ambient water on coral polyps. Starting with the first mechanism, most often reported in the scientific literature, numerous experiments performed to date on tropical reef-building corals suggest that when coral species are exposed to the increased CO2 concentrations, their calcification rates are significantly being reduced (Marubuni and Atkinson 1999; Kleypas et al. 2006; Fischlin et al. 2007).

Marubuni and Atkinson (1999) investigated the relative effects of different pH levels on growth of corals, using coral tips of the hermatypic coral *Porites compressa*. Researchers reported that corals growing in seawater at a reduced pH level of 7.2 calcified at half the rate of control corals growing at pH level of 8.0. However, researchers pointed out that corals which were a subject to low pH treatments recovered their initial calcification rates within 2 days of re-introduction to ambient seawater, indicating the effects of CO2 chemistry are immediate and reversible.

Fine and Tchernov (2007) investigated the impact of the pH changes in the ambient water on 30 coral fragments from five coral colonies of the scleractinian Mediterranean species *Oculina patagonica* (encrusting) and *Madracis pharencis* (bulbous), which were exposed to two different ranges of pH: lower pH levels ranging from 7.3 to 7.6 and the ambient (control) pH levels ranging from 8.0 to 8.3, for the period of 12 months. After 1 month in acidic conditions, morphological changes were observed, ranging from polyp elongation, followed by dissociation of the colony form and complete skeleton dissolution. Surprisingly, the polyps remained attached to the undissolved hard rocky substrate. Control and treatment coral fragments maintained their algal symbionts during the entire experiment, except for 10% of *O. patagonica* that partially bleached, but recovered within 2 months. Researchers reported that all skeleton-free coral fragments survived to the end of the experiment and after 12 months, when returned to the ambient pH conditions, polyps calcified and rebuilt colonies. This study demonstrates that skeleton-producing corals grown in acidified experimental conditions are able to sustain basic life functions, including reproductive ability, in a sea anemone-like form and will resume skeleton building when reintroduced to normal modern marine conditions, mainly implying that corals might survive large-scale environmental change, such as that expected for the following century.

Changes in pH of the ambient water can also potentially impact the settlement rates of new coral recruits. This mechanism was documented by research done by Kuffner (2007) on crustose coralline algae, which form the structural crust on reef flats and attract settlement of new coral recruits. Because these species of algae are responsible for "cementing" carbonate frameworks, they are particularly vulnerable to reduced growth and recruitment rates from ocean acidification (Kuffner 2007). This research supports findings of other researchers who also reported that impacts of ocean acidification on crustose coralline algae will likely negatively affect corals by reducing coral settlement rates (Eakin et al. 2009).

The potential reduction in the presence and diversity of the corals present at sites with lower pH levels is the another mechanism reported in the literature. Fabricius et al.(2011) reported that with decreasing pH the marine waters, the number and types of corals building coral reefs are much reduced. Researchers further report that the diversity of corals in Papua New Guinea, dropped by 40% and the reef became dominated by one form of corals, massive boulder corals (Porites). The cover of the more delicate branching corals was reduced three-fold near the CO_2 seeps. Similarly, the abundance of soft corals and sponges were also significantly reduced. Overall, Fabricius et al. found that reef development ceased below pH level 7.7

Anthony et al. (2008) evaluated effects of different pH levels on growth rates of Pacific Acropora species. Researchers reported that as a result of the intermediate-CO_2 dosing (resulting in pH from 7.85 to 7.95) a 50% reduction in productivity was observed, relative to the control. At the same time, the high-CO_2 dosing (resulting in pH 7.60 to 7.70) led to a further reduction in productivity to near zero. This research suggested that the ocean acidification may reduce the threshold at which coral bleaching occurs; however, both elkhorn and staghorn corals have yet to be subjected to the similar acidification studies.

In the research study published by Kaniewska et al., the specimens of reef building *Acropora millepora* species were subjected to decreased pH (Kaniewska et al. 2012) for 28 days. Control specimens were held in the seawater with pH ranging from 8 to 8.2 (corresponding to pCO2 concentration ranging from 260 to 440 ppm). The medium CO_2 treatment was controlled to a pH ranging from 7.8 to 7.9 (corresponding to pCO2 concentration ranging from 600 to 790 ppm) and the high CO_2 treatment was targeted to a pH ranging from 7.6 to 7.7 (corresponding to pCO2 concentration ranging from 1010 to1350 ppm). Changes in the metabolism, calcification rates, and cellular activities of specimens were observed with different pH exposure. In addition to the increased difficulty to form hard skeletons at lower pH levels, the energy invested into the process of calcification was observed to be "much more costly." As it was previously described, corals rely on zooxanthellae for energy generated via process of photosynthesis. Kaniewska et al. showed that increased level of CO_2 in the ambient water caused coral branches of *A.millepora* to lose their symbiotic algae.

The second important finding of Kaniewska et al. study was the observation of significant increase in internal cellular pH regulation by the *A. millepora* corals due to changes in CO_2 levels in the ambient water. This increase in the internal pH regulation has a potential to result in less energy being devoted to calcification. By decreasing calcification, not only does ocean acidification decrease coral growth, but it also decreases the accretion of the reef system as a whole. In summary, Kaniewska et al. also reported that, as pH levels decrease in the ambient water, some corals are capable of further up-regulating their internal pH, in order to continue

calcifying in acidic conditions present in the ambient waters. As a result of this ability, some corals may exhibit less sensitivity to pH changes than others. The loss of zooxanthellae to stress or bleaching events, however, would significantly reduce the effectiveness of this ability. Kaniewska et al. stressed the need to expand future studies of ocean acidification on corals to include a wider spectrum of cellular processes, many of which may occur before impacts on calcification.

Increased ocean acidification was also reported to have significant impact on *Acropora formosa* corals by reducing the photosynthetic capacity and photoprotection of their symbiotic algae (Crawley et al. 2010). When *A. formosa* was exposed to increased CO2 levels, the production of a key enzyme that protected its symbiotic algae from sunlight was significantly reduced, which exposed the algae to oxidative stress and reduced their ability to convert sunlight into nutrients for the coral.

Meron at al. (2011) examined changes in bacterial communities in the coral mucus, living tissue and skeleton following 10-week exposure of the coral *Acropora eurystoma* to two different pH conditions: 7.3 and 8.2 (ambient seawater). As a result, the microbial community was different at the two pH levels. Further analysis of the community in the corals maintained at the lower pH revealed an increase in bacteria associated with diseased and stressed corals. In addition, an increase in the number of potential antibacterial activity was recorded among the bacteria isolated from the coral maintained at pH 7.3. This study clearly highlighted yet another impact that changes in the pH may have on the coral-associated bacterial community and their potential contribution to the coral host.

The final listing rule for seven of our ESA-listed coral species identified elevated CO2 as a threat that may be contributing to the status of our coral species of concern (NMFS 2006 and 2014). However, the severity of ocean acidification impacts on the individual coral species is unknown While it was a disease and not the acidification what caused the initial decline of our seven ESA-listed coral species, the severity of this threat to their growth, fertilization success, and recruitment will make it more difficult for them to recover from the historically low populations currently present. As summarized above, there are numerous cases, being documented in the scientific literature, reporting coral degradation caused by changes in water quality due to ocean acidification.

As previously mentioned, there are still many gaps in the knowledge of impacts of pH changes in the ambient water on the coral recovery, in general. It will take time to assess the long term response of corals to ocean acidification in combination with other environmental stresses. In the meantime, reduced calcification and slower growth rates will potentially result in their slower recovery from breakage, whether natural (hurricanes and storms) or human caused (breakage from vessel groundings, anchors, fishing gear, etc.), or mortality from a variety of disturbances. Slower growth rates will require more time for young coral polyps to reach reproductive size. It will also result in higher mortality rates for newly settled corals. This is to be expected because young corals are in general more vulnerable to overgrowth competition, sediment smothering, and incidental predation until they reach a refuge at larger colony size.

In summary, as a result of the literature review, the EPA did not locate any scientific information directly related to the impacts of changes in the ambient pH on the recovery of any of the ESA-

listed coral species with the exception of two species within *Acropora* genus (*A. palmata* and *A. cervicornis)* and the limited study on *Orbicella faveolata* specimen. However, multiple research was found related to the other species within the Acropora genus: *millepora, formosa* and *eurystoma*. The genus Acropora represents over 149 stony coral species, which share many similar characteristics. They all are colonies of individual polyps building the calcium carbonate sub-structures, are most common in shallow reef environments with bright light and moderate to high water motion (Richards et al. 2008). As a result, for the purpose of this consultation, the EPA considered both *A. palmata* (Elkhorn coral) and *A. cervicornis* (Staghorn coral) species to be as sensitive to pH levels in the ambient waters as other three Acropora species evaluated above.

In addition, the EPA was not able to locate any scientific information directly related to the pH impacts on the remaining four newly listed coral species of concern. Because three-quarters of reef-building (stony) coral species was reported to spawn gametes and rely on external fertilization and planktonic development, Albright et al. (2010) considered elkhorn coral *A. palmata* to be a representative of spawning species in general. The EPA agrees with this approach, and evaluated characteristics for all three Genus: Dendrogyra, Orbicella and Mycetophyllia and their anatomic as well as habitat similarities. The EPA did not come across any information precluding the Agency from assuming that five newly listed coral species will be protected by the pH criterion adopted by the USVI to the same degree as two Acropora corals.

During the literature review process, the EPA came across multiple reports recommending specific pH ranges, as optimal conditions for the individual coral species, when setting up the residential aquaria. The EPA did not find any scientific information either supporting the recommendations or suggesting the optimal pH levels for ambient marine waters to protect the individual coral species in their natural habitat. As it was already previously indicated, there were many gaps identified in the research which need to be filled before the most optimal pH conditions can be derived for the individual coral species and their habitats around the USVI.

Based on the research results summarized above, the EPA acknowledges that the lower ranges of the pH criteria (pH of 6.7 or 7.0), adopted by the USVI for Class B and C waters, may not be fully protective of ESA-listed coral species or their habitats. However, based on the fact that along with the pH ranges, the USVI also adopted the provision stating that at any time, the natural range of pH in the ambient water must not be extended at any location by more than +/- 0.1 pH unit, the agency believes that this additional "natural" provision provides adequate protection for the ESA-listed species.

Conclusion: The USVI has adopted a pH level that cannot be altered except of natural conditions for Class A waters; pH level of not less than 7.0 or greater than 8.3 for Class B waters and pH level of not less than 6.7 or greater than 8.5 for Class C waters. For Classes B and C, the USVI adopted an additional provision stating that the normal (natural) range of pH must not be extended at any location by more than +/- 0.1 pH unit.

In summary, EPA determined that pH criterion adopted by the USVI for Class A waters represents the natural pH conditions existing in the ambient marine waters. As a result, this narrative criterion is deemed to be fully protective of the ESA-listed coral species and their habitats, thus it is determined NLAA ESA-listed coral species or their critical habitats. EPA

plans to work with the VIDPNR and NMFS to reevaluate the existing pH criteria and revise them, if necessary, to ensure that they are fully protective of all ESA-listed coral species and their habitats around the USVI. The revisions to pH criteria will be considered during the next VIWQSR review process scheduled for 2018.

b) Analysis of potential impacts of pH on Whales

The information of impacts of pH levels present in the marine waters on whale species is very limited. Based on the scientific literature review, many researchers are presently looking into potential changes in pH levels predicted overtime due to ocean acidification. There is no information available in the literature on the optimal pH levels or on the direct impact of pH levels present in the ambient water on condition of ESA-listed whales. However, in general, researchers expect that the predicted pH changes will over time will result in changes to whale's habitats and shifts in the spatial distribution of their prey.

Whales are highly migratory and as such can easily relocate to the most optimal water conditions, at any given time. In addition, whales have tendency to dive into deep waters, thus their bodies are adjusted to sudden changes in water quality occurring in various water depths. Although pH levels in marine waters are expected to decrease overtime, there is no information being presently published in the scientific literature, indicating potential pH changes to be a direct threat to ESA-listed species of whales.

A Sperm whale, *Physeter macrocephalus,* is expected to be less sensitive to indirect impact of pH levels, compared to the remaining four ESA-listed species. This whale is a toothed whale and does not relay on the plankton as a source of food.

- Blue whale, *Balaenoptera musculus;* Fin whale, *Balaenoptera physalus;* Sei whale, *Balaenoptera borealis* and Humpback whale, *Megaptera novaeangliae*

The remaining four ESA-listed whale species belong to the baleen group of filter-feeders. Laboratory studies suggest that some of the oceanic plankton species are highly sensitive to changes in CO2 concentrations in sea water. The calcification rate of all calcifying organisms investigated to date decreased in response to a decreased calcium carbonate saturation state (Feely et al. 2004). Calcifying organisms that may be affected include the coccolithophores, pteropods, gastropods and foraminifera; all of which are major food sources for baleen whale species.

There are no optimal pH conditions identified in the scientific literature for any of the whale species, however, the literature suggests that, in general, the ocean productivity will likely to be affected by changes in climate and corresponding changes in the marine environment (Mackas *et al.* 1989; Quinn and Niebauer 1995). In the Final Recovery Plan for Fin whales, NMFS indicated that currently used habitat areas may become unsuitable due to expected changes in water quality related to climate change. However, the primary threats identified by NMFS

included changes in the ocean currents and water temperature, and were not related to changes in pH. Research indicates that changes to climate and oceanographic processes may also lead to decreased productivity in different patterns of prey distribution and availability. Such changes could affect baleen whales that are dependent on those affected prey. For example, distribution of copepods has already shown signs of shifting in the North Atlantic due to increasing CO2 concentrations in the ambient waters (Hays *et al.* 2005).

In summary, the potential changes in ambient pH levels present in marine waters and their impact on wellbeing of whales, remain unknown. The uncertainty of threat posed by environmental variability to the recovery of ESA-listed whales was ranked high by NOAA, due to the unknown potential impacts of climate and ecosystem changes on whale's recovery and regime shifts of their prey. Overall, the relative impact of the environmental variability to whales recovery was ranked as unknown (NMFS 2010).

Conclusion: The USVI adopted water quality criteria for pH of no less than 7.0 or greater than 8.3 for Class A and B marine waters and no less than 6.7 or greater than 8.5 for Class C marine waters. There is no information in the scientific literature suggesting that the pH criteria adopted by the USVI are not protective and potentially pose a threat to the survival and recovery of ESA-listed whale species. Taking into consideration all of the above information, EPA has determined that the pH criteria adopted for all of VI's marine waters are beneficial to marine environment and result in full protection of all five species of whales and their habitats. As a result, EPA considers the U.S.V.I pH criteria NLAA Sperm, Fin, Sei, Blue and Hampback whales or their critical habitats.

c) Analysis of potential impacts of pH on Sea Turtles

The EPA did not find in the literature any information which would indicate the pH range adopted by the USVI as water quality criterion to be a direct threat to the recovery of the ESA-listed sea turtles. Similarly to the effects of DO, discussed previously, the most significant (indirect) effect of pH levels of ambient water on sea turtles is potential degradation of their habitats as well as the availability and distribution of their prey.

- Hawksbill sea turtle, *Eretmochelys imbricata* and Green sea turtle, *Chelonia mydas*

Coral reef habitats are very sensitive to changes in water quality, which is especially of concern for two of the ESA-listed sea turtle species, which are either frequent or regular visitors to the coral reef ecosystems: Hawksbill turtle, which feeds in the lagoon or back reef zone of coral reef ecosystems and the Green turtle, which feeds primarily on the seagrasses found in protected back reef lagoons. The impacts of ambient concentrations of pH on sensitive coral reef ecosystems were briefly described in the earlier section of this document. Hawksbill turtles are mainly found on and around coral reefs feeding primarily on sponges. As a result, this species is strongly dependent of the high quality waters required for this habitat to survive. Green turtles are

herbivores, as adults forage among seagrass beds and nearshore habitats feeding primarily on algae, seagrasses, and seaweed (hatchlings are omnivores). The seagrass habitats are also very sensitive to water quality changes, however with focus on changes to water temperature and turbidity rather than pH.

- Loggerhead sea turtle, *Caretta caretta* and Leatherback sea turtle, *Dermochelys coriacea*

Loggerhead turtles are carnivores (adults), eating crabs, conchs, whelks, and horseshoe crabs. Leatherbacks, which are deep divers, are found in pelagic (open ocean) environments where they feed exclusively on jellyfish and other soft-bodied invertebrates that float in the water column (tunicates and sea squirts). The leatherback has been tracked crossing the entire Pacific Ocean from Asia to the US West coast to forage on swarms of jellyfish off the coasts. Although lower pH is preferable for this species, rapid changes in pH can be very dangerous to jellyfish. Scientists report that the number of jellyfish is on the rise thanks to the increasing acidity of the world's oceans.

Conclusion: The USVI adopted water quality criteria for pH of no less than 7.0 or greater than 8.3 for Class A and B marine waters and no less than 6.7 or greater than 8.5 for Class C marine waters. There is no information in the scientific literature suggesting that the pH criteria adopted by the USVI are not protective and potentially pose a threat to the survival and recovery of ESA-listed sea turtle species. Taking into consideration all of the above information, EPA has determined that the pH criteria adopted for all of VI's marine waters are beneficial to marine environment and result in full protection of four ESA-listed sea turtle species and their habitats. As a result, EPA considers the U.S.V.I pH criteria NLAA Green, Hawksbill, Loggerhead and Leatherback turtles or their critical habitats.

d) Analysis of potential impacts of pH on Fish

- Scalloped Hammerhead Shark, *Sphyrna lewini*

During the scientific literature review, the EPA did not come across any research which reported on the specific effects of pH on Scalloped Hammerhead Shark. In addition, the EPA did not find any information which suggested that the pH range adopted by the USVI as water quality criterion would pose a direct threat to the recovery of this species.

Conclusion: The USVI adopted water quality criteria for pH of no less than 7.0 or greater than 8.3 for Class A and B marine waters and no less than 6.7 or greater than 8.5 for Class C marine waters. There is no information in the scientific literature suggesting that the pH criteria adopted by the USVI are not protective and potentially pose a threat to the survival and recovery of Scalloped Hammerhead Shark. Taking into consideration all of the above information, EPA has determined that the pH criteria adopted for all of VI's marine waters are beneficial to marine environment and result in full protection of Scalloped Hammerhead Shark and its habitat.

(please refer to the corals section above for details). As a result, EPA considers the U.S.V.I pH criteria NLAA Scalloped Hammerhead Shark or its critical habitat.

Should additional information related to the effects of pH on Scalloped Hammerhead Shark becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

- Nassau Grouper, *Epinephelus striatus*

There is no information in the literature regarding specific pH ranges that are optimal for this species. There is also no information in the literature suggesting that the pH range adopted by the USVI is a direct threat to the recovery of this species. Taking into consideration that the Nassau grouper lives around coral reefs, it is anticipated that the most likely (indirect) effect of the pH on this species will be degradation of its habitat. The effect of pH on coral reef ecosystem was described in the earlier sections of this BE.

Conclusion: The USVI adopted water quality criteria for pH of no less than 7.0 or greater than 8.3 for Class A and B marine waters and no less than 6.7 or greater than 8.5 for Class C marine waters. There is no information in the scientific literature suggesting that the pH criteria adopted by the USVI are not protective and potentially pose a threat to the survival and recovery of Nassau Grouper. As a result, EPA considers the U.S.V.I pH criteria NLAA Nassau Grouper species. The determination related to the habitat of this species is the same as the EPA determination for the pH criteria for coral reef protection described earlier in the document.

Should additional information related to the effects of pH on Nassau Grouper becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

3. Water Temperature

The temperature of seawater varies with the amount of sun that hits the given area, the latitude of the location and its depth. Tropical areas that get more year-round sun and more direct sun have warmer surface waters compared to polar areas, resulting in warm surface ocean temperatures in the tropics (up to 30 degrees Celsius [^{o}C] or more) and cooler at the poles (down to -2 ^{o}C). The temperature of seawater also varies with depth. Oceans are vertically stratified and marine scientists recognize a basic three layered ocean: the upper mixed layer, up to 200 m (where temperature depends on latitude and the season); the main thermocline, from 200 m to 1000 m (where temperature rapidly decreases with depth) and deep (bottom) water (where temperature stays between -2 to 5 ^{o}C). Every life form within the marine system has its thermal range most optimal for its survival. Climate-related changes in ocean temperature have been identified by

NMFS as one of the major environmental threats to the marine aquatic life. The surface ocean temperature is projected to increase by approximately 0.7 °C by 2030 and 1.4 °C by 2060 compared to the 1986-2005 average, with the 10 to 90% range increasing over that time period to approximately +/-0.7 °C by 2060 (IPCC 2013). Based on the results of the literature review, it appears that for many coral species, a one-time increase of only 1 °C to 2 °C above the normal local seasonal maximum ocean temperature can induce bleaching. Deeper areas are generally less affected typically because lower irradiance reduces the likelihood of warming-induced bleaching, thus making corals located at higher depths less sensitive to ocean warming.

a) Analysis of potential impacts of temperature on Corals

As reported by Eakin et al. (2009), reef-building coral species live within a fairly narrow range of environmental conditions and are highly sensitive to changes in water temperature, light, salinity, nutrients, bathymetry and the aragonite saturation state of seawater. All seven of the ESA-listed coral species are zooxanthellate corals, which relay on endosymbiotic dinoflagellates for energy and growth. Zooxanthellae have a mutualistic symbiotic relationship with their hermatypic hosts; in return for habitat and nutrients, the zooxanthellae provide photochemically useable energy for the coral, thus accounting for a large portion of a coral's net energy (Knowlton and Rohwer 2003). Hoegh-Guldberg (1999) reported that the zooxanthellate corals are predominantly located in tropical coastal waters with temperatures ranging from 18 to 30 °C. When ocean temperatures exceed summer maximum by 1to 2 °C for 3 to 4 weeks, zooxanthellate corals expel their endosymbiotic algae, resulting in weaker and less able to combat diseases. This process is known as bleaching (Hoegh-Guldberg et al. 2007). Shinn (1966) reported that staghorn coral expelled zooxanthellae at or near 33 °C.

Overall, the researchers are in general agreement on the temperature range tolerated by the stony corals. The minimum sea surface temperature to which functional reefs are normally exposed to was reported to be 18 ^{o}C since pioneer investigations done by Dana in 1843, through many decades including the 1984 publication by Rosen (Veron 1995). This minimum temperature threshold was reevaluated by Veron and Minchin in 1992 and it was found to hold true.

- Elkhorn coral, *Acropora palmata*

In the Final Listing Rule, NMFS (2014) concluded that *A. palmata* is highly susceptible to ocean warming. Lundgren and Hillis-Starr (2008) reported *A. palmata* colonies to be sensitive to bleaching. In Trunk Bay and Saltpond, St. John, U.S. Virgin Islands, almost half of the colonies that bleached in 2005 suffered partial or complete mortality (44% of 27 colonies and 40% of 107 colonies, respectively. In St. Croix, U.S. Virgin Islands, colonies differentially bleached in Buck Island National Monument during the 2005 Caribbean-wide mass bleaching event; colonies in the shallower back reef bleached earlier and suffered greater tissue loss than those located elsewhere (Lundgren and Hillis-Starr, 2008).

High water temperatures also affect *A. palmata* reproduction. *A. palmata* embryos and larvae exhibited more developmental abnormalities, lower survivorship, and decreased settlement at 30 and 31.5°C compared to those at 28 °C (Randall and Szmant, 2009). Larvae of *A. palmata* exhibited faster development and faster swimming speed at 30 and 31.5 °C compared to controls at 27 and 28 °C (Baums et al., 2013).

- Staghorn coral, *Acropora cervicornis*

In the Final Listing Rule, NMFS (2014) concluded that *Acropor cervicornis* is highly susceptible to ocean warming. As reported by NOAA, this species is considered to be highly susceptible to bleaching in comparison to other coral species, with a variable rate of mortality. Waddell and Clarke (2008) reported approximately 75% of *A. cervicornis* colonies to bleach at 12 monitored sites and 90% of the *A. cervicornis* colonies to show a partial or total mortality during and after the 2005 bleaching event in Puerto Rico.

A. cervicornis was one of the most heavily affected species during a 1987 to 1988 bleaching event in the Cayman Islands with 100% of colonies bleached on the deep reef terrace (18 to 29 m depth) and 83% bleached on the shallow reef terrace (Ghiold and Smith, 1990). In Roatan, Honduras, Riegl et al. (2009) monitored *A. cervicornis* and found none were bleached fully during the 1998 bleaching event, with the fourth highest partial bleaching frequency, and the highest mortality of 22 species monitored. During the 2005 bleaching event with 17 species observed, only *A. cervicornis* and *A. palmata* bleached 100% (all colonies bleached completely white) at two reefs in Jamaica with 90 % mortality at one site and 10% at the other .

Van Woesik et al. (2012) developed a coral resiliency index based on biological traits and processes to evaluate extinction risk due to bleaching. Evaluations were performed at the genus level with genera separated between the Caribbean and Indo- Pacific. They reported *A. cervicornis* as highly vulnerable to extinction due to bleaching.

- Pillar coral, *Dendrogyra cylindrus*

In the Final Listing Rule, NMFS (2014) concluded that although *Dendrogyra cylindrus* appears to have resistance to bleaching from warmer temperatures in some portions of its range under some circumstances, it is likely to have some susceptibility to ocean warming.

There are conflicting characterizations of the susceptibility of *D. cylindrus* to bleaching due to water warming. Some locations experienced high bleaching of up to 100% of *D. cylindrus* colonies during the 2005 Caribbean bleaching event while others had a smaller proportion of colonies bleach (10 to 50%).

D. cylindrus appears to be sensitive to cold temperatures. In laboratory studies of cold shock, *D. cylindrus* had the highest zooxanthellae expulsion rate of three species tested at 12 °C (Muscatine et al. 1991). During the 2010 cold water event in the Florida Keys, *D. cylindrus* was

one of the most affected coral species with 100% mortality on surveyed inshore reefs (Kemp et al. 2011).

- Lobed star coral, *Orbicella annularis*

In the Final Listing Rule, NMFS (2014) concluded that *Orbicella annularis* is highly susceptible to ocean warming.

The *O. annularis* species complex is sensitive to cold water. In laboratory experiments, *O. annularis* species complex released zooxanthellae when shocked with cold water between 12 and 18 °C, and the response decreased with increasing temperature (Muscatine et al., 1991).

Surveys from 19 locations throughout the Caribbean indicated the bleaching event of 1995-96 was most extensive in the central and western Caribbean but only slight in the Lesser Antilles and Bermuda. Mortality of *O. annularis* from bleaching ranged from 2 to 30% at eight locations six months after the onset of bleaching (Alcolado 2003).

Extended recovery times have been reported, and disease outbreaks have often followed bleaching events. Bleaching often occurs in 76 to 94% of *O. annularis* species complex colonies during bleaching events, and *Orbicella spp*. are one of the taxa most affected by high temperatures. Colonies in deeper water have been reported to bleach less severely. Recovery from bleaching can take longer for the species complex than for other coral species, and prolonged stress from bleaching has been cited as a possible reason for reproductive failure following bleaching events. Bleaching has been shown to prevent reproduction in the following season after recovering normal pigmentation Mortality from temperature anomalies is often due to subsequent disease outbreaks. A significant correlation was found between bleaching in 2005 and the prevalence of yellow band disease and white plague affecting the Orbicella species complex. Additionally, in laboratory experiments, mortality due to yellow band disease increased with increasing temperatures.

- Mountainous star coral, *Orbicella faveolata*

In the Final Listing Rule, NMFS (2014) concluded that *Orbicella faveolata is* highly susceptible to ocean warming.

As described by NOAA (2014), *O. faveolata* is highly susceptible to elevated temperatures. In lab experiments, elevated temperatures resulted in misshapen embryos and differential gene expression in larvae that could indicate negative effects on larval development and survival. Bleaching susceptibility is generally high with 37 to 100 % of *O. faveolata* colonies reported to bleach during several bleaching events. Chronic local stressors can exacerbate the effects of warming temperatures, which can result in slower recovery from bleaching, reduced calcification, and slower growth rates for several years following bleaching. Additionally, disease outbreaks affecting *O. faveolata* have been linked to elevated temperature as they have occurred after bleaching events.

Stratified random surveys on back-reefs and fore-reefs between one and 30 m depth off Puerto Rico in 2005 and 2006 revealed severe bleaching in *O. faveolata* with approximately 90% of colonies bleached (Waddell and Clarke, 2008). Surveys from 2005 to 2007 along the Florida reeftract indicated that *O. faveolata* had the13th highest bleaching prevalence out of 30 species observed to bleach (Wagner et al., 2010). During a 2009 bleaching event on Little Cayman, of the ten coral species that bleached, *O. faveolata* had the third highest bleaching prevalence with approximately 37% of colonies bleached (van Hooidonk et al. 2012).

Experiments exposing *O. faveolata* to high temperatures (up to 35 °C) revealed that the corals produced heat shock proteins at temperatures between 33 and 35 °C, even for very short exposures (2 hours) but did respond at temperatures between 27 and 31 °C when exposed from 2 hours to one week (Black et al. 1995).

Voolstra et al. (2009) exposed *O. faveolata* embryos to temperatures of 27.5, 29, and 31.5 °C directly after fertilization and measured differences in gene expression after 12 and 48 hours. They found a higher number of misshapen embryos after 12 hours at 29 and 31.5 °C in comparison to embryos kept at 27.5 °C . However, after 48 hours, the proportion of misshapen embryos decreased for embryos kept at 29 and 31.5 °C, and increased for embryos kept at 27.5 °C.

- Boulder star coral, *Orbicella franksi*

In the Final Listing Rule, NMFS (2014) concluded that *Orbicella franksi* is highly susceptible to ocean warming.

Available information indicates that *O. franksi* is highly susceptible to warming temperatures with a reported 88 to 90 % bleaching frequency. Reported bleaching-related mortality from one study is high at 75%. There is indication that symbiont shuffling after bleaching in *O. franksi.*

During a bleaching event in Colombia in 2010, 88% of *O. franksi* bleached, and 12% paled at a site in Gayraca Bay (Bayraktarov et al. 2012). In 2011, 75% of *O. franksi* were dead and completely overgrown by algae. Based on sample s of *O. franksi* and *O. faveolata* collected from the Mesoamerican Barrier Reef, calcification of these two species is projected to cease at 35 °C in this location in the absence of acidification .

Stratified random surveys on back-reefs and fore-reefs between one and 30 m depth off Puerto Rico (Mona and Desecho Islands, La Parguera, Mayaguez, Boqueron, and Rincon) in 2005 and 2006 revealed severe bleaching in *O. franksi* with approximately 90% of colonies bleached (Waddell and Clarke, 2008). Surveys from 2005 to 2007 along the Florida reef indicated *O. franksi* had the tenth highest bleaching prevalence out of 30 species observed to bleach (Wagner et al., 2010).

- Rough cactus coral, *Mycetophyllia ferox*

In the Final Listing Rule, NMFS (2014) concluded that *M. ferox* has some susceptibility to ocean warming. However, the available information does not support a more precise description of susceptibility to this threat.

In surveys of the lower Florida Keys and Dry Tortugas during the 1998 bleaching event, approximately % of *M. ferox* colonies bleached; out of the 14 species reported to have experienced bleaching of at least 50% of the colony, *M. ferox* was one of the least affected (Waddell, 2005). Approximately 50 % of *M. ferox* colonies bleached at 12 locations in Puerto Rico during the 2005 bleaching event (Waddell and Clarke, 2008). During the 2005 Caribbean bleaching event, neither of the two colonies of *M. ferox* monitored at six sites in Barbados bleached; an average of 71 % of all coral colonies bleached at those six sites during the event .

The bleaching reports available specifically for *M. ferox* and at the genus level indicate similar trends of relatively low bleaching observed in 1995, 1998, and 2010 (less than 25 %) and higher levels (50 to 65) or no bleaching in the more severe 2005 bleaching event. Reproductive failure and a disease outbreak were reported for the genus after the 2005 bleaching event. Although bleaching of most coral species is spatially and temporally variable, understanding the susceptibility of *M. ferox* is somewhat confounded by the species' low sample size in any given survey due to its low encounter rate.

In addition to the species-specific studies, the EPA came across additional information which was very relevant to the determination of the most optimal temperature conditions for the recovery of coral species, in general. Based on current studies, temperature of ambient water was identified as a key factor for the long-term coral survival. Research suggests that coral reefs are unable to withstand high temperatures, which in most cases cause bleaching. This situation was observed in Singapore during the 1998 coral bleaching event, whereby an increase in 1 to 2 °C in the sea temperatures around Pulau Hantu and St John's islands affected 50-90% of all reef organisms in Singapore. Research further indicates that photosynthesis pathways in zooxanthallae become impaired at temperatures above 30 °C, activating the disassociation of coral/algal symbiosis (Buchheim 1998). These findings are consistent with the general understanding that coral's survival above temperatures of 30 °C or above is questionable and that the optimum temperature for healthy coral growth range from 28 to 29 °C (Wells 1957, Stoddart 1962). The National Marine Sanctuary Habitats website reported the tolerated by corals ambient water temperatures ranging from 61 to 95 degrees Fahrenheit [°F] (16 to 35 °C), with the most optimal temperature window of 73 to 77 °F (23 to 25 °C).

Temperature of the ambient water was shown to affect the metabolic rates of corals and their symbionts (Coles and Brown, 2003). This study indicated that corals are adapted to their ambient temperature conditions. For example, photosynthesis to respiration rations for the same species of corals in Hawaii and Enewetak, across a temperature ranging from 18 to 31 ^{o}C, indicated that Enewetak corals were adapted to their higher ambient temperatures. Coles and Brown (2003) also reported that there is no specific temperature threshold which would cause coral bleaching in general, however, the prolonged exposure of corals to increases of 1 to 3 ^{o}C above long-term annual maximum temperatures (ranging from 25 ^{o}C to 35-36 ^{o}C) are likely to induce bleaching.

Overall, the exposure to temperatures exceeding the tolerance range of the symbiosis was shown to affect its stability, generally resulting in the loss of symbiotic algae and possibly in the death of the coral host. Whether the loss is due to a direct temperature effect on the coral, algae, or both, remains unclear however, the rate of the temperature change and the duration of the temperature anomaly appear to be factors.

Another example of sensitivity of corals to changes in temperature was published by Miller et al. (2009). Miller at al. presented the results of the elevated water temperatures, which created the most severe coral bleaching event ever documented within the USVI region. As a result of temperature changes ranging from 0.7 to 1.2 ^{o}C higher than the historical average reported for this area, resulted in over 90% of the scleractinian coral cover showing signs of thermal stress by paling or becoming completely white. Lower water temperatures in following months allowed some re-coloring of corals; however, a subsequent unprecedented regional outbreak of coral disease affected all sites. Miller et al. suggested that corals recovering from bleaching are likely to be more susceptible to diseases for several months.

The above findings are in the agreement with the suggested temperature range provided by NOAA (http://oceanservice.noaa.gov/facts/coralwaters.html), which indicates that reef-building corals cannot tolerate water temperatures below 64° F (18° C) and many grow optimally in water temperatures between 73° and 84° F (23°– 29 °C). NOAA's website also indicated that Mayer (1914) reported that the lethal temperature for elkhorn coral was between 34 and 35°C and the decreased larval survival and settlement of elkhorn coral have been found at ocean temperatures above 30 °C (Randall and Szmant 2009). Therefore, mean monthly sea surface temperatures likely will need to be below 30 °C during spawning periods to improve successful coral reproduction.

In summary, although the EPA did not locate any scientific information directly related to the optimal temperature conditions specified for the individual ESA-listed coral species of concern, the Agency considers the temperature range of 25 to 29 °C, identified in the literature by multiple researchers as the most optimal, to be equally protective temperature threshold for all seven ESA-listed coral species of concern.

Conclusion: The USVI adopted a temperature criterion of not to exceed 32 °C at any time, applicable to Class B and C waters and the additional, more stringent, criterion of not to exceed 25-29 °C at any time, for the entire Class A waters and areas within Class B and C waters where coral reefs are known to be located. In all cases, the temperature resulting from any waste discharge must not be greater than 1 °C above natural. For the additional protection, the USVI also adopted thermal policies, which are applicable to all water.

The scientific literature review by the EPA suggests that the temperature criteria adopted by the USVI are protective and do not pose a threat to the survival of ESA-listed coral species. Taking into consideration all of the information reported in the scientific literature summarized above, the EPA has determined that the temperature criteria adopted by the USVI are beneficial to marine environment and result in full protection of seven ESA-listed coral species and their

habitats. As a result, EPA considers the USVI's temperature criteria (NLAA ESA-listed coral species or their critical habitats.

b) Analysis of potential impacts of temperature on Whales

There is no information in the literature indicating the optimal temperature conditions for whales. Indirect impacts of the water temperature are expected to be related to changes in whale's habitats and shifts in the spatial distribution of their prey (for both, baleen and tooth whales). With changing climate, water temperatures are expected to increase over time. The specific effects of changes in ocean temperature on whales' habitat and distribution of their prey is yet to be examined.

In general, whales are highly migratory and as such can easily relocate to the most optimal water conditions. In addition, whales have tendency to dive into deep waters, thus their bodies are adjusted to sudden temperature changes occurring at different ocean depths.

- Sperm whale, *Physeter macrocephalus*

Sperm whale is a toothed whale and feeds on large prey such as large squid and fish, including some species of sharks. Because the distribution range of sperm whales is extensive, this species is expected to be more resilient to climate change (thus, increased ocean temperature) than a species with a narrower distribution range. In general, the optimal water temperature for whales is the one which is optimal for their prey. Giant squid comprise about 80% of the sperm whale diet and has a high oxygen demand (Guerra et al. 2011). The remaining 20% of sperm whale diet is comprised of octopus, fish, shrimp, crab and even small bottom-living sharks (NOAA retrieved at: http://www.afsc.noaa.gov/nmml/education/cetaceans/sperm.php.). The optimal temperature preferences for this open-ocean and deep-ocean prey greatly vary. Squid, the primary prey of sperm whales, may be negatively impacted by rising ocean temperatures, especially in the Antarctic. Krill quantities are predicted to decrease and, as the primary prey of squid, this may have implications for squid populations. However, squid are opportunistic feeders, and they may be able to adapt to changes in krill abundance by feeding on a variety of organisms. In addition, it is worth noting that the feeding range of sperm whales is likely the greatest of any species on earth, and, consequently, it's likely that sperm whales will be more resilient to climate change than species with a narrow range of habitat preferences.

- Blue whale, *Balaenoptera musculus;* Fin whale, *Balaenoptera physalus;* Sei whale, *Balaenoptera borealis* and Humpback whale, *Megaptera novaeangliae*

Unlike sperm whale, the remaining four ESA-listed whale species belong to baleen group of filter-feeders. Blue whales eat mostly krill. Fin whales eat krill, copepods, squids, and variety of small schooling fishes. Humpback whales prey mostly on krill and small schooling fishes. Sei

whales eat copepods, krill and amphipods (another type of small crustacean). Krill quantities are predicted to decrease with rising temperature of water. Over time, increase water temperatures are expected to impact dense patches of zooplankton. In such cases, female whales would have a difficult time to prepare for calving, carry a pregnancy to term or produce enough milk. When the concentration of zooplankton is too low, whales do not feed. Changes of seawater temperature (along with winds and water currents) can potentially affect patch formation of zooplankton The loss of sea ice in Antarctica has already caused a decrease in the amount of algae, plankton and krill, the foundation of the ocean's food chain.

The USVI adopted the temperature criterion of not to exceed 32 ℃ at any time, applicable to Class B and C waters and the additional, more stringent, criterion of not to exceed 25-29 ℃ at any time, for the entire Class A waters and areas within Class B and C waters where coral reefs are known to be located. In all cases, the temperature resulting from the waste discharge must not be greater than 1 ℃ above natural. For the additional protection, the USVI also adopted thermal policies, which are applicable to all water.

Conclusion: There is no information in the scientific literature that would suggest that the temperature criteria adopted by the USVI are not protective and potentially pose a threat to the recovery and survival of the five ESA-listed whale species. Taking into consideration all of the above information, EPA has determined that the USVI temperature criteria, are beneficial to marine environment and result in full protection of whales and their habitats. As a result, EPA considers the U.S.V.I temperature provisions NLAA Sperm, Blue, and Fin, Sei and Humpback whales or their critical habitats.

Should additional information related to the effects of temperature on whale speces becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

c) Analysis of potential impacts of temperature on Sea Turtles

Sea turtle biologists are beginning to assess how changes in major ocean currents, key habitats, weather patterns, and prey abundance and distribution resulting from climate change and ocean acidification will affect sea turtles and their habitats (Sea Turtles Conservancy accessed at: http://www.conserveturtles.org/velador.php?page=velart90). Climate change is expected to alter numerous habitats utilized by sea turtles including beaches, sea grass beds, coral reefs, near-shore bottom areas as well as the waters of the open ocean. For the purpose of this BE, the EPA will focus on the evaluation of the temperature of water as it relates to the aquatic ecosystems rather than nesting beaches, which remain under the jurisdiction of the US FWS.

The seagrasses on which green turtles depend are expected to become less productive as a result of warmer water (along with the increased sedimentation and runoff from coastal flooding, and decreased visibility and light penetration). In the coral reefs with which hawksbills are closely associated, increased water temperatures (along with acidification) will undermine the ability of

coral polyps to survive and build reefs. Other threats include variation in prey abundance and range.

- Hawksbill sea turtle, *Eretmochelys imbricata* and Green sea turtle, *Chelonia mydas*

Coral reef habitats are very sensitive to changes in water quality, which is especially of concern for two of the ESA-listed sea turtle species, which are either frequent or regular visitors to the coral reef ecosystems: Hawksbill turtle, which feeds in the lagoon or back reef zone of coral reef ecosystems and the Green turtle, which feeds primarily on the seagrasses found in protected back reef lagoons. The impacts of ambient water temperature on sensitive coral reef ecosystems were briefly described in the earlier section of this document. Hawksbill turtles are mainly found on and around coral reefs feeding primarily on sponges. As a result, this species is strongly dependent of the high quality waters required for this habitat to survive. Green turtles are herbivores, as adults forage among seagrass beds and nearshore habitats feeding primarily on algae, seagrasses, and seaweed (hatchlings are omnivores). The seagrass habitats are also very sensitive to water quality changes, especially changes in temperature. Overall, sea grass beds are in decline, water temperature is higher on intertidal sea grass flats, and coral reefs, typically feeding grounds for green turtles, are affected by bleaching.

- Loggerhead sea turtle, *Caretta caretta* and Leatherback sea turtle, *Dermochelys coriacea*

ESA-listed sea turtles face threats on both nesting beaches and in the marine environment. According to NOAA (http://www.nmfs.noaa.gov/pr/species/turtles/loggerhead.htm), major threats include the degradation or loss of nesting and foraging habitats. Negative impacts of increased temperature are well documented in the literature in relation to the beach environment. Rising temperatures increase the chance that sand temperature will exceed the upper limit for egg incubation (34°C). Rising temperatures also bias the sex ratio toward females because temperature during incubation determines the sex of the egg. Loggerhead turtle nests in Florida are already producing 90% females owing to high temperatures. There is no scientific literature indicating the optimal water temperature for any of the ESA-listed turtle species. There is also no indication that the temperature range adopted by the USVI as the water quality standard poses a direct threat to the recovery and survival of any of the ESA listed species.

The USVI adopted the temperature criterion of not to exceed 32 °C at any time, applicable to Class B and C waters and the additional, more stringent, criterion of not to exceed 25-29 °C at any time, for the entire Class A waters and areas within Class B and C waters where coral reefs are known to be located. In all cases, the temperature resulting from the waste discharge must not be greater than 1 °C above natural. For the additional protection, the USVI also adopted thermal policies, which are applicable to all water.

Conclusion: There is no information in the scientific literature that would suggest that the temperature criteria adopted by the USVI are not protective and potentially pose a threat to the recovery and survival of the four ESA-listed sea turtle species. Taking into consideration all of

the above information, EPA has determined that the USVI temperature criteria, are beneficial to marine environment and result in full protection of sea turtles and their habitats. As a result, EPA considers the U.S.V.I temperature provisions NLAA Hawksbill, Green, Loggerhead and Leatherback turtles or their critical habitats.

Should additional information related to the effects of temperature on sea turtle species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

d) Analysis of potential impacts of temperature on Fish

- Scalloped Hammerhead Shark, *Sphyrna lewini*

During the scientific literature review, the EPA did not come across any research which would report on the specific effects of water temperature on Scalloped Hammerhead Shark. In addition, the EPA did not find any information which would suggest that the temperature range adopted by the USVI as water quality criterion would pose a direct threat to the recovery of this species.

- Nassau Grouper, *Epinephelus striatus*

Increased water temperature due to the climate change has its implication on Nassau grouper survival (NOAA 2010). This species have been found across a range of temperatures, however spawning occurs only when sea surface temperatures are approximately 25°C. If sea surface temperatures rise, the geographic range of the species is expected to shift. One of the other potential effects of climate change could relate to the loss of structural habitat in the coral reef ecosystems (Munday et al. 2008). Increased sea surface temperatures have been responsible for coral loss through bleaching and disease reducing adult habitat for Nassau grouper (Coleman and Koenig 2010).

The USVI adopted the temperature criterion of not to exceed 32 °C at any time, applicable to Class B and C waters and the additional, more stringent, criterion of not to exceed 25-29 °C at any time, for the entire Class A waters and areas within Class B and C waters where coral reefs are known to be located. In all cases, the temperature resulting from the waste discharge must not be greater than 1 °C above natural. For the additional protection, the USVI also adopted thermal policies, which are applicable to all water.

Conclusion: There is no information in the scientific literature that would suggest that the temperature criteria adopted by the USVI are not protective and potentially pose a threat to the recovery and survival of both ESA-listed fish species. Taking into consideration all of the above information, EPA has determined that the USVI temperature criteria, are beneficial to marine environment and result in full protection of Nassau grouper and Scalloped Hammerhead Shark and their habitats. As a result, EPA considers the U.S.V.I temperature provisions NLAA both ESA-listed fish species or their critical habitats.

Should additional information related to the effects of temperature on fish species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

4. Turbidity

Turbidity is a measure of the amount of suspended particulate matter in the water column (and to a lesser extent some dissolved organic compounds) and their effect on light attenuation. Both organic (bacteria, phytoplankton, zooplankton and detritus), or inorganic (sediment) particles contribute to the suspended particulate matter. Turbidity and light attenuation can vary over small spatial and temporal scales depending on the proximity of sources of sedimentation and/or terrestrial runoff as well as changes in local weather conditions. Sedimentation has been identifies by NOAA-NMFS as a key stressor for the recovery of many ESA-listed species and their habitats. Increased sediments often accompany nutrients and chemical contaminants from terrestrial runoff, coastal erosion, resuspension of bottom sediments, beach nourishment, nearshore dredging and coastal development/construction projects.

Turbidity and light availability in the marine environment are measured and expressed in a number of different ways (Erftemeijer 2012). Common measures for turbidity include concentration of total suspended solids (TSS, in mg/L), suspended-sediment concentration (SSC, in mg/L), nephelometric turbidity units (in NTU), Secchi disc readings (in cm), and attenuation coefficient (k_d). Conversion factors between these different measures are site-specific, depending on various local factors, including particle-size distribution, contribution of phytoplankton and organic content (Thackston and Palermo 2000).

In order to roughly compare the TSS concentrations reported in the scientific literature in mg/L units to the turbidity criterion adopted by the USVI in NTU units, for the purpose of this Biological Evaluation, the EPA used the laboratory derived correlation curve published in the technical note by Thackston and Palermo (2000). Using the laboratory derived relationship between turbidity values measured in the NTU units and the TSS values measured in mg/L units, authors derived the laboratory correlation curve to be able to estimate turbidity levels (in NTUs) to represent TSS concentrations (in mg/L). In their technical note, Thackston and Palermo reported that their laboratory generated correlation curve estimated a turbidity of 40 NTUs to represent a TSS of 70 mg/L (with a range of 55 to 90 mg/L). On the other hand, the correlation curve generated based on the field data (using sediment samples obtained from Mobile Harbor area) estimated a turbidity of 40 NTUs to represent a TSS of 52 mg/L (with a range of 25 to 60 mg/L). While this correlation was derived specifically for Mobile Harbor sediment samples, due to the lack of correlation curves derived for sediment samples collected in the USVI, the EPA considers the laboratory derived correlation curve to be an acceptable tool to estimate how the turbidity criteria of “not to exceed 1 or 3 NTUs" adopted by the USVI compare to the TSS values reported in the scientific literature as protective of ESA-listed species. At the same time, the Agency realizes that the use of the above described correlation curve results only in "approximate" estimates rather than more exact calculations.

a) Analysis of potential impacts of turbidity on Corals

- Elkhorn coral, *Acropora palmata*

In the Final Listing Rule, NMFS (2014) concluded that *A. palmata* is highly susceptible to sedimentation.

Acropora palmata is sensitive to sedimentation due to its poor capability of removing sediment and its high reliance on clear water for nutrition, and sedimentation can cause tissue mortality. The morphology of *A. palmata* contributes to its sensitivity to sedimentation as it is poorer at removing sediment compared to mounding corals such as Orbicella annularis and Diploria strigosa (Abdel-Salam et al. 1988). Out of five species tested, *A. palmata* was the least tolerant of sediment exposure; single applications of 200 mg/cm2 to colonies caused coral tissue death as sediments accumulated on the flattened, horizontal surfaces (Rogers 1983). Because *A. palmata* is highly dependent on sunlight for nutrition, it is also sensitive to suspended sediments that reduce water clarity (Porter 1976). In Vega Baja, Puerto Rico, *A. palmata* mortality increased to 52% concurrent with pollution and sedimentation associated with raw sewage and beach nourishment, respectively, between December 2008 and June 2009 (Hernandez-Delgado et al. 2010).

- Staghorn coral, *Acropora cervicornis;*

In the Final Listing Rule, NOAA-NMFS (2014) concluded that *A. cervicornis* is highly susceptible to sedimentation.

Acropora cervicornis is sensitive to turbidity because it is highly reliant on sunlight for nutrition (Porter, 1976). Rogers (1979) shaded an area of 20 meter sq. of reef as a partial simulation of conditions from turbidity and found that *A. cervicornis* was the first species to respond by bleaching. Three weeks after shading was initiated, most colonies of *A. cervicornis* were bleached. After shading was terminated at five weeks, at the sixth week, most branches were dead and covered with algae with growth tips deteriorating or grazed away, but a few branches recovered. After seven weeks, there were more algae on the branches and further disintegration of branch tips.

Acropora cervicornis is susceptible to sedimentation through its sensitivity to turbidity, and increased run-off from land clearing has resulted in mortality of this species. In addition, laboratory studies indicate the combination of sedimentation and nutrient enrichment appears to be worse than the effects of either of these two stressors alone.

- Lobed star coral, *Orbicella annularis*

In the Final Listing Rule, NMFS (2014) concluded that *Orbicella annularis* is highly susceptible to sedimentation.

As reported by Nugues and Roberts (2003), in St. Lucia, rates of partial mortality of *O. annularis* and *O. faveolata* were higher close to river mouths where sediments were deposited than they were farther from the river mouths, indicating the sensitivity of these two species to sedimentation. Sedimentation can cause partial mortality of *O. faveolata*, and genus-level information indicates that sedimentation negatively affects primary production, growth rates, calcification, colony size, and abundance.

Orbicella has shown a decline in growth at sediment impacted sites in Puerto Rico and during periods of construction in Aruba. Along a gradient of continental influence in the southern Gulf of Mexico, density and calcification rate of *O. annularis* decreased with increasing turbidity and sedimentation while extension rate increased with increasing turbidity and sedimentation. *O. annularis* species complex appears to be moderately capable of removing sediment from the colony surface. Colonies receiving single applications of 200 or 400 mg sediment per cm2 showed no evidence of damage while 800 mg per cm2 caused mortality (Rogers, 1983).

Sedimentation has been found to negatively affect *O. annularis* species complex primary production, growth rates, and abundance (Pastorok and Bilyard, 1985). An observed difference in average colony size at two sites in Puerto Rico led Loya (1976) to conclude turbidity negatively affects growth of *O. annularis* species complex since colony size was half as large at the sediment-impacted site (23 cm versus 9 cm).

- Pillar coral, *Dendrogyra cylindrus;* listed in 2014

In the Final Listing Rule, NMFS (2014) concluded that *Dendrogyra cylindrus* has some susceptibility to sedimentation, but the available information does not support a more precise description of susceptibility to this threat...

D. cylindrus appears to be moderately capable of removing sediment from its tissue. However, *D. cylindrus* may be more sensitive to turbidity due to its high reliance on nutrition from photosynthesis and as evidenced by the geologic record of susceptibility to this threat. In addition to the species-specific information summarized above, the EPA came across literature which provides more general information on impacts of turbidity on different coral species.

The primary source of sediments affecting near-shore coral reefs is the terrestrial runoff resulting from the coastal development and dredging processes, which usually result in a sediment plume which may settle onto corals adjacent to or downstream from the construction/dredged area. The severity of the increased turbidity in the ambient water (due to sedimentation) on the recovery of the ESA-listed coral species depends on the type of sediments and the size and the hydrodynamic conditions of the impacted site, as well as the duration of sedimentation process. Each of these factors influences the size, settlement time, and ultimate settling site of the sediment plume.

Sediment particles smother reef organisms and reduce light available for photosynthesis (Rogers 1990). Excessive sedimentation can adversely affect the structure and function of the coral reef ecosystem by altering both physical and biological processes. Rogers reported mean sediment rates and suspended sediment concentrations for reefs not subject to stresses from human activities to be less than 1 to 10 mg/cm2/day and less than 10 mg/ L, respectively, which is significantly higher to the turbidity criteria adopted by the USVI. Chronic rates and concentrations above suggested by Rogers values were considered to be high. In general, heavy sedimentation is associated with fewer coral species, less live coral, lower coral growth rates, greater abundance of branching forms, reduced coral recruitment, decreased calcification, decreased net productivity of corals, and slower rates of reef accretion.

Turbidity and light attenuation can have contrasting effects on corals. While some species gain a substantial proportion of their energy budgets from heterotrophic feeding on suspended particulate matter (usually ones located in deep waters), while others obtain most of their nutrition from autotrophy (symbiotic zooxanthellae provide the host coral with sugars, amino acids, lipids and peptides) regardless of the availability of particulate matter. The latter mechanism is used by all seven of our ESA-listed coral species of interest.

Because the zooxanthellae corals depend on phototrophic nutrition and photosynthetically enhanced calcification, these coral species prefer, and are found in the shallow, clear waters. Total daily production depends on the amount of light penetrating water depth, transmitted through the coral animal tissues and available to algae. Photosynthetic rates increase directly in response to increase in light intensity up to a certain light intensity, after which they are either independent of further increases in light or inhibited by high light (Hatcher 1988). In general, the upper layers of the coral canopy receive more light compared to the understory surfaces of coral branches and as a result, zooxanthellae located in shaded and unshaded portions of the colony may exhibit different degrees of photoadaptation.

In case of reef-building corals, which rely on symbiotic zooxanthellae for energy generated through the process of photosynthesis, the primary problems arising from turbidity and sedimentation are related to the shading caused by decreases in ambient light and sediment cover on the coral's surface. In addition, turbidity and sedimentation is also causing problems for the feeding apparatus under a sediment blanket and energetic costs associated with mucus production, sediment clearance and impaired feeding. Suspended sediments, especially when fine-grained, decrease the quality and quantity of incident light levels, resulting in a decline in photosynthetic productivity of zooxanthellae (Falkowski et al. 1990).

The research data suggest that coral reproduction and recruitment are far more sensitive to changes in water quality than adult corals (Fabricius 2005). Sediment deposition and accumulation was reported to result is loss of suitable substrate available for larval settlement, recruitment, and fragment reattachment (Babcock and Davies 1991, Birrell et al. 2005), where both sediment composition and deposition, affected the survival of juvenile corals resulting in the increased larval mortality (Fabricius 2005). The accumulation of sediments was reported to smother coral recruits (Babcock and Davies 1991, Fabricius 2005). In addition, settlement rates for coral larvae, and reattachment rates for fragments, were reported to be near zero on sediment-covered surfaces, and sedimentation tolerance in coral recruits is at least one order of magnitude lower than for adult corals (Fabricius 2005).

Overall, the increased sedimentation caused smothering and burial of coral polyps, shading, tissue necrosis and significantly increased levels of bacteria in coral mucus. In general, sediments are associated with lower coral species richness and abundance, lower growth rates, decreased calcification, decreased net productivity, and lower rates of coral recruitment (Rogers 1990). In addition, coral species are reported to have a different capabilities of clearing themselves of sediment particles or surviving lower light levels (Rogers 1990). Sediment rejection is a function of morphology, orientation, growth habit, and behavior; and of the amount and type of sediment.

Based on the reviewed literature, it appears that the elkhorn and staghorn corals are particularly sensitive to sediment deposition and shading effects from increased sediment regimes. Both species require relatively clear, well-circulated water and are highly dependent upon sunlight for nourishment (Porter 1976). Both elkhorn and staghorn corals have poor capacity to remove coarser sediments (250-2000 μm) and only slightly more capacity for removing finer sediments (62-250 μm) (Hubbard and Pocock 1972). Water movement (turbulence) and gravity are probably more important in removing sediments from these species than their capabilities of sloughing sediments in still water (Porter 1976).

Erftemeijer et al. (2012) publication provides a review of published literature related to the sensitivity of corals to turbidity and sedimentation, with an emphasis on the effects of dredging. In general, this literature review indicates that the risks and severity of impact from sediment disturbances on corals are primarily related to the intensity, duration and frequency of their exposure to increased turbidity and sedimentation. The information presented by Erftemeijer et al. was related to 89 different coral species, however only 10% of those represented known reef-building corals, which are a subject of this evaluation. In general, the duration of high turbidities which corals survived ranged from several days (sensitive species) to at least 5–6 weeks (tolerant species). Erftemeijer et al. (2012) reported a range of sensitivities to turbidity for many coral species, including two of our ESA-listed species of interest. For both, staghorn coral (*A. cervicornis)* and lobed star coral (*M. annularis)*, after 4-day exposure: no effect was observed with concentration of 50 mg/L (approx. 29 NTUs); partial mortality was reported with concentration of 476 mg/L (approx. 272 NTUs), and the complete mortality was reported with concentration of 1000 mg/L (approx. 571 NTUs).

Rogers (1983) investigated the effects of sedimentation on three of our coral species of interest: staghorn coral, elkhorn coral, and lobed star coral. Elkhorn coral was the least tolerant of sediment deposition, with coral tissue death reported after exposure to the sedimentation rate of 200 mg/cm2 (most likely due to the deposition of sediment on the "flat" horizontal portions of the coral structure). Although the staghorn coral colonies (with widely spaced, cylindrical branches) were found by Rogers to be more tolerant to the same sediment deposition (most likely due to the passive sediment removal), Hodel and Vargas-Ángel (2007) noted degenerative histopathological changes in staghorn coral exposed to the same sedimentation rates, indicating sub-lethal damage to the coral tissue impacting the overall health of the colony.

Flores et al. (2012) exposed two species of coral to six levels of total suspended solids (TSS) for 16 weeks in the laboratory, including a 4 week recovery period. Dose-response relationships were developed to quantify the lethal and sub-lethal thresholds of sedimentation and turbidity for the corals. As a result, the sediment treatments affected the horizontal foliaceous species

(*Montipora aequituberculata*) more than the upright branching species (*Acropora millepora*). The lowest sediment treatments that caused full colony mortality were TSS concentrations of 30 mg/L TSS (approx. 17 NTUs) (25 mg/cm2/ day) for *M. aequituberculata* and TSS of 100 mg/L (approx. 57 NTUs) (83 mg/cm2/day) for *A. millepora* after 12 weeks. Coral mortality generally took longer than 4 weeks and was closely related to sediment accumulation on the surface of the corals.

Flores et al. reported that the exposure to sediments can produce a range of different responses in corals. Although feeding on fine sediment particles was found to enhance coral growth in some species (Anthony 1999), in general, however, settling of particulate matter onto the colony surface is considered a stress to corals due to down-regulation of photosynthesis and increased rates of respiration and mucous production (Telesnicki and Goldberg 1995). Photo-physiological stress occurs within hours of exposure to sedimentation and is strongly related to grain size, organic content and nutrient composition of the sediment. With increasing exposure to sediments, coral growth rates decline, symbionts are known to be expelled (bleaching), and tissue loss occurs (Miller and Cruise 1995). Sedimentation also negatively affects rates of gamete fertilization and survival and settlement of coral larvae (Babcock and Davies 1991). In the longer term, elevated sedimentation regimes can influence coral cover and community composition due to differences in sediment tolerances among species.

In general, Flores et al. indicated that the levels of sedimentation and turbidity impacts on corals vary according to species, polyp size and growth form. In general, corals are thought to be affected by chronic sediment deposition rates greater than 10 mg/cm2/day and TSS above 10 mg/L (resulting in approx. 5 NTUs), but this is highly dependent on sediment properties (corals have the greatest difficulty in expelling and removing the finest sediment fractions).

The final listing rule (NMFS 2006a) identified sedimentation as a threat contributing to the threatened status of elkhorn and staghorn corals. Similarly, the final rule maintaining the two species as threatened (NMFS 2014) lists sedimentation as a threat contributing to their status because of their susceptibility to this threat, similarly to the decision made for the remaining five ESA-listed coral species of interest. The steep island topography of Puerto Rico and the USVI increases the sediment loads in terrestrial run-off, which increases the exposure to sediment accumulation on the surrounding coral reefs. Thus, in these territories, the threat of sedimentation is ranked medium (3) for staghorn corals and high (4) for elkhorn corals due to their differing morphology.

In general, sediments are known to negatively impact coral reef ecosystems; however, there is a significant gap in the information allowing researchers to derive protective threshold levels and link the sediment deposits to their impacts on the specific ESA-listed species and their habitats. As a result, the EPA was not able to locate any scientific information directly related to the optimal turbidity levels for any of the ESA-listed species. However, taking into consideration that the background and the lowest turbidity treatment levels evaluated in the research summarized above were significantly higher that the levels adopted by the USVI as water quality criteria, the EPA considers the existing USVI turbidity standard to be fully protective of all ESA-listed species under investigation.

Conclusions: The USVI adopted a maximum permissible NTU reading of three (3) for Class B and C waters and a maximum permissible NTU reading of one (1) applicable to all Class A waters and locations within Class B and C waters, where coral reef ecosystems are known to be located.

Taking into consideration all of the information reported in the scientific literature summarized above, the EPA has determined that the turbidity criteria adopted by the USVI are beneficial to marine environment and result in full protection of seven ESA-listed coral species and their habitats. As a result, EPA considers the USVI's turbidity criteria NLAA ESA-listed coral species or their critical habitats.

b) Analysis of potential impacts of turbidity on Whales

- Sperm whale, *Physeter macrocephalus;* Blue whale, *Balaenoptera musculus;* Fin whale, *Balaenoptera physalus;* Sei whale, *Balaenoptera borealis* and Humpback whale, *Megaptera novaeangliae*

During the scientific literature review, the EPA did not come across any research reported on the specific effects of water turbidity on ESA-listed whales. In addition, the EPA did not find any information which would suggest that the turbidity level adopted by the USVI as water quality criterion would pose a direct threat to the recovery or survival of this species.

Conclusions: The USVI adopted a maximum permissible NTU reading of three (3) for Class B and C waters and the maximum permissible NTU reading of one (1) applicable to all Class A waters and locations within Class B and C waters, where coral reef ecosystems are known to be located. The EPA has determined that turbidity criteria adopted by the USVI are beneficial to marine environment and result in full protection of seven ESA-listed whale species and their habitats. As a result, EPA considers state's turbidity criteria NLAA ESA-listed whale species or their critical habitats.

Should additional information related to the effects of turbidity on whale species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

c) Analysis of potential impacts of turbidity on Sea Turtles

- Hawksbill sea turtle, *Eretmochelys imbricata* and Green sea turtle, *Chelonia mydas*; Loggerhead sea turtle, *Caretta caretta* and Leatherback sea turtle, *Dermochelys coriacea*

There is no information in the scientific literature indicating the optimal turbidity levels for any of the ESA-listed turtle species. There is also no indication that the turbidity levels adopted by the USVI pose a direct threat to the recovery or survival of any of ESA-listed species. However, the indirect effect of water turbidity is expected to play important role on the condition of their habitats. It is well documented in the literature that the coral reef and seagrass habitats are especially sensitive to turbidity and that water of high clarity is a key to their survival. The turbidity levels, as they relate to the most sensitive coral ecosystems were already evaluated earlier in this document and the determination was made of their protectiveness for all ESA-listed coral species and their habitats. As a result, the EPA believes that the same turbidity levels will be as protective for the remaining habitats of the ESA-listed species of sea turtles.

Conclusions: The USVI adopted the maximum permissible NTU reading of three (3) for Class B and C waters and the maximum permissible NTU reading of one (1) applicable to all Class A waters and locations within Class B and C waters, where coral reef ecosystems are known to be located. The EPA has determined that turbidity criteria adopted by the USVI are beneficial to marine environment and result in full protection of ESA-listed sea turtle species and their habitats. As a result, EPA considers state's turbidity criteria NLAA ESA-listed sea turtle species or their critical habitats.

Should additional information related to the effects of turbidity on sea turtle species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

d) Analysis of potential impacts of turbidity on Fish

- Scalloped Hammerhead Shark, *Sphyrna lewini*

During the scientific literature review, the EPA did not come across any research which would report on the specific effects of water turbidity on Scalloped Hammerhead Shark or its habitat. In addition, the EPA did not find any information which would suggest that the turbidity level adopted by the USVI as water quality criterion would pose a direct threat to the recovery or survival of this species.

- Nassau Grouper, *Epinephelus striatus*

Decline of tropical fisheries is partially attributable to deterioration of coral reefs, seagrass beds, and mangroves from sedimentation and increased turbidity (Rogers 1990). Turbid waters often alter the complex interactions between fish and their reef habitat. For example, elevated turbidity of water can destroy major reef-building corals, leading to a decline in the amount of

shelter the reef provides and to reductions in both number of individuals and number of species of fish.

It is well documented in the literature that the coral reef and seagrass habitats are especially sensitive to the turbid waters and that water of high clarity is a key to their survival. The turbidity levels, as they relate to the most sensitive coral ecosystems were already evaluated earlier in this document and the determination was made of their protectiveness for all ESA-listed coral species and their habitats. As a result, they are also protective of Nassau Grouper.

Conclusions: The USVI adopted a maximum permissible NTU reading of three (3) for Class B and C waters and the maximum permissible NTU reading of one (1) applicable to all Class A waters and locations within Class B and C waters, where coral reef ecosystems are known to be located. The EPA has determined that turbidity criteria adopted by the USVI are beneficial to marine environment and result in full protection of ESA-listed fish species and their habitats. As a result, EPA considers state's turbidity criteria NLAA Scalloped Hammerhead Shark and Nassau Grouper as well as their critical habitats.

5. Clarity

Water clarity measures the clearness (transparency) of ambient water, indicating how far sunlight is able to travel through the water column. Water clarity is measured by the Secchi depth and is closely related to water turbidity, where increased turbidity correlates with decreased clarity. Transparency is often being used as an indicator of water turbidity since transparency can be affected by the water color and the amount of suspended materials present in the water.

Clarity is an important water quality parameter for plants that depend on light interception and absorption for photosynthesis and the animals depending on these plants. Water clarity measurement indicates the depth of the photic zone; the zone of water that is exposed to enough sunlight to support photosynthesis. The depth of the photic zone varies with the turbidity of the water.

In the process of the scientific literature review, the EPA did not come across any research providing the recommendations for the optimal water clarity conditions for the recovery of any of the ESA-listed species of corals, whales, see turtles and fish. Unlike the turbidity, water clarity was not identified by NOAA-NMFS in the listing process, as a potential threat for the recovery of any of our seventeen ESA-listed species of interest.

The EPA would like to point out that the water quality parameters affected by low water clarity are, for purposes of this Biological Evaluation, the same as parameters affected by the high water turbidity, which were already addressed earlier in this document. High water clarity, as much as low water turbidity, is a key factor for a well-being of the marine ecosystems. Reduced water clarity, as much as increased water turbidity, leads to prolonged shading and limits depth distribution of coral reefs and seagrasses. In addition, it leads to reduced coral biodiversity and increased macroalgal cover. The intensity of the light penetrating water column was also found

to greatly affect photosynthetic rates of the zooxanthellae, indirectly impacting coral growth and survival.

Conclusion: The USVI has adopted a clarity criterion of a minimum of 1 meter for a secchi disc to be visible in all marine Classes of water (A, B and C). Taking into consideration the nature of clarity measurement described above, the water clarity criterion of 1 meter does not ensure that the sunlight will be able to penetrate the water column and reach zooxanthellae coral reefs located at higher depths, allowing for the photosythesis process to take place. As a result, the EPA acknowledges that clarity criterion adopted by the USVI may not be fully protective of all ESA-listed corals species located around the USVI. However, taking into consideration that all of the ESA-listed coral species are already being fully protected by the turbidity criteria adopted by the USVI, the EPA believes that all of the negative effects potentially caused by the low clarity in the water column are very unlikely to occur at any given location where the stringent turbidity criteria is being met.

As a result, EPA has concluded its evaluation of water clarity criteria with NLAA determination. This determination is based on the best judgment that the potential effects of not fully protective water clarity criterion, are considered to be discountable (thus extremely unlikely to occur).

EPA plans to work with the VIDPNR and NOAA-NMFS to reevaluate the existing water clarity criterion and revise it, to ensure that it is fully protective of all ESA-listed coral species and their habitats around the USVI. The revisions to water clarity criterion will be considered during the next VIWQSR review process scheduled for 2018.

6. Phosphorous

a) Analysis of potential impacts of phosphorus on Corals

Nutrient enrichment is considered under ESA Factor A – the present or threatened destruction, modification, or curtailment of its habitat or range – and ESA Factor E – other natural or manmade factors affecting the continued existence of the species – because the effect of the threat, resulting from human activity, is both to limit the availability of habitat for corals and directly impact individuals of coral species. In the proposed rule we described the threat of nutrient over-enrichment as follows. Elevated nutrients affect corals through two main mechanisms: direct impacts on coral physiology and indirect effects through nutrient-stimulation of other community components (e.g., macroalgal turfs and seaweeds, and filter feeders) that compete with corals for space on the reef. Increased nutrients can decrease calcification; however, nutrients may also enhance linear extension, while reducing skeletal density.

Nutrients are largely recognized as elements that are beneficial for marine organisms. Sources of nutrients include anthropogenic outlets, such as sewage and stormwater discharges and urban and farm runoff (Szmant 2002) as well as coastal aquaculture activities. Naturally occurring sources include leaf litter (Szmant 2002), excretion of digested planktonic biomass by sponges,

and tidal upwelling events (Leichter et al 2003). Marine ecosystems, especially coral reefs, are adapted to low nutrient levels, and overabundance of nutrients can result in an imbalance that affects the entire ecosystem. Excess nutrient loads have been shown to affect coral physiology and the balance between corals and their endosymbiotic zooxanthellae (Szmant 2002). Nutrient-rich water can enhance benthic algae and phytoplankton growth rates in coastal areas, and this may result in overgrowth, outcompetition, and algal blooms. Increased levels of nutrients have also been shown to reduce growth rates in staghorn corals and compromise their health.

Nutrients are added to coral reefs from both point sources (e.g. readily identifiable inputs where pollutants are discharged to receiving waters from a single source such as a pipe or drain) and non-point sources (inputs that occur over a wide area and are associated with particular land uses). Anthropogenic sources of nutrients include sewage, stormwater, and agricultural runoff, river and inlet discharge, and groundwater. Natural oceanographic sources like internal waves and upwelling also deliver nutrients to coral reefs. Coral reefs generally have been considered nutrient-limited systems, meaning that levels of accessible nitrogen and phosphorus limit the rates of macroalgae growth. When nutrient levels are raised in such a system, growth rates of fleshy macroalgae are expected to increase. Whether this increase in productivity translates into higher abundance of macroalgae on reefs depends on the level of herbivory removing that biomass (Szmant 2002).

- Elkhorn coral, *Acropora palmata*

In the Final Listing Rule, NMFS (2014) concluded that *Acropora palmata* is highly susceptible to nutrient enrichment.

In Vega Baja, Puerto Rico, *A. palmata* mortality increased to 52% concurrent with pollution and sedimentation associated with raw sewage and beach nourishment, respectively, between December 2008 and June 2009 (Hernandez-Delgado et al. 2011). Mortality presented as patchy necrosis-like and white pox-like conditions that impacted local reefs following anthropogenic disturbances and was higher inside the shallow platform (52 to 69%) and closer to the source of pollution (81 to 97%) compared to the outer reef (34 to 37%) (Hernandez-Delgado et al. 2011). *Acropora palmata* was reported as sensitive to nutrients as evidenced by increased mortality after exposure to raw sewage.

- Staghorn coral, *Acropora cervicornis*

In the Final Listing Rule, NMFS (2014) concluded that *Acropora cervicornis* is highly susceptible to nutrient enrichment.

Renegar and Riegl (2005) performed laboratory experiments to examine the effect of nutrients and CO2 on growth of *A. cervicornis* branch tips, maintained in the laboratory. Researchers measured coral growth before, during, and after exposure to elevated nitrate (5 and 10 µM), phosphate (2 and 4 µM) and/or CO2 (approx. approximately 700 to 800 µatm). Researchers

reported significantly reduced growth under CO2 levels of 700 to 800 µatm, predicted to occur this century, compared to controls. In addition, when elevated CO2 was combined with increased nitrate and phosphate, growth rates were further reduced. The effect of combined nitrate, phosphate, and CO2 appeared to be antagonistic, at lower nutrient concentrations and additive, at higher concentrations (compared to those nutrients paired with CO2 separately). Growth rate recovery was greater after exposure to increased nutrients or CO2 compared to increased nutrients and CO2. All corals in the combined nitrate, phosphate, and CO2 treatment experienced total mortality, indicating the severe stress this combination induced. Under the nutrient treatments alone, *A. cervicornis* experienced significantly lower growth rates under the higher nitrate and higher phosphate treatments, though not under the lower levels, and the combined nitrate and phosphate treatment produced significantly lower growth under both the low and high levels.

Laboratory experiments testing the effects of sedimentation and phosphate on *A. cervicornis* indicated that degenerative changes to tissue, zooxanthellae, and gonad development were more severe in sediment plus phosphate treatments in comparison to controls and phosphate alone (Hodel and Vargas-Angel, 2007).

- Pillar coral, *Dendrogyra cylindrus*

In the Final Listing Rule, NMFS (2014) concluded that *D. cylindrus* likely has some susceptibility to nutrient enrichment. However, the available information does not support a more precise description of its susceptibility to this threat.

- Lobed star coral, *Orbicella annularis*

In the Final Listing Rule, NMFS (2014) concluded that *Orbicella annularis* is highly susceptible to nutrient enrichment.

Elevated nutrients cause increased disease severity in *O. annularis*. Genus level information indicates elevated nutrients also cause reduced growth rates and lowered recruitment.

Field experiments indicate that nutrient enrichment significantly increases yellow band disease severity *in O. annularis* and *O. franksi* through increased tissue loss (Bruno et al. 2003). In laboratory experiments, dissolved organic carbon caused significantly higher mortality of *O. annularis* after 30 days of exposure compared to controls while nutrients (phosphate, nitrate, and ammonia) did not (Kline et al. 2006; Kuntz et al. 2005).

Dissolved organic carbon levels that resulted in significantly higher mortality compared to controls were 12.5 mg /L glucose, and 25 mg /L lactose, starch, galactose, and glucose, which were all levels reported for impacted reefs (Kline et al. 2006; Kuntz et al. 2005).

- Mountainous star coral, *Orbicella faveolata*

In the Final Listing Rule, NMFS (2014) concluded that *Orbicella faveolata is* likely highly susceptible to nutrient enrichment.

The *O. annularis* species complex is susceptible to nutrient enrichment through reduced growth rates, lowered recruitment, and increased disease severity. The *O. annularis* species complex is susceptible to nutrient enrichment through reduced growth rates, lowered recruitment, and increased disease severity.

Although there is no species-specific information for this species, Orbicella genus, in general, is susceptible to nutrient enrichment through reduced growth rates, lowered recruitment, and increased disease severity (likely *O. faveolata*) and lobate (likely *O. annularis*) were found to have increasing average growth rates with improving environmental conditions away from a eutrophication gradient in Barbados (Tomascik 1990).

Although nutrient concentration was negatively correlated with growth, suspended particulate matter resulting from eutrophication, rather than the nutrients themselves, was postulated to be the cause of observed decreased growth rates (Tomascik and Sander 1985). A general pattern of decreasing growth rates of the columnar growth form between 1950 and 1983 may be directly related to the deterioration of water quality along the west coast of the island (Tomascik 1990). Additionally, Orbicella spp. did not recruit to settlement plates on the most eutrophic reef, and recruitment of Orbicella spp. increased at sites with decreasing eutrophication along the eutrophication gradient (Tomascik 1991).

- Boulder star coral, *Orbicella franksi*

In the Final Listing Rule, NMFS (2014) concluded that *Orbicella franksi* is likely highly susceptible to nutrient enrichment.

Field experiments indicate that nutrient enrichment significantly increases yellow band disease severity in *O. annularis* and *O. franksi* through increased tissue loss (Bruno et al., 2003).All sources of information are used to describe *O. franksi's* susceptibility to nutrient enrichment as follows. Genus level information indicates *O. franksi* is likely susceptible to nutrient enrichment through reduced growth rates and lower recruitment. Additionally, nutrient enrichment has been shown to increase the severity of yellow band disease in *O. franksi*.

- Rough cactus coral, *Mycetophyllia ferox*

In the Final Listing Rule, NMFS (2014) concluded that *M. ferox* likely has some susceptibility to nutrient enrichment. However, the available information does not support a more precise description of susceptibility. As noted by NOAA, *M. ferox* may be susceptible to nutrient enrichment as evidenced by its absence from eutrophic sites. However, there is uncertainty

about whether the absence is a result of eutrophic conditions or a result of uncommon or rare occurrence of that species.

In addition to the species-specific data, the EPA obtained more general information on impacts of phosphorus on coral species, in general. Corals are known to thrive in seawater where the concentrations of nitrogen (N) and phosphorus (P), are very low (Szmant et al. 1990). In the case of zooxanthellae corals, scientists believe that algal-coral symbiosis evolved in response to this relatively low ambient nutrient concentration, providing corals with a competitive advantage over other benthic species. Although corals are well adapted to waters containing very low levels of nutrients, corals are known to persist when nutrient levels around reefs become periodically elevated (due to increased run-off, sewage and industrial effluents). However, corals are known to be unable to adapt to acute, high level nutrient enrichments, especially over the longer periods of time.

There are various sources of nutrients identified in the literature for symbiotic zooxanthellae. This type of coral species developed two specific mechanisms to obtain nutrition in generally low-nutrients environment. In the first case, corals can capture zooplankton by polyps and utilize translocated photosynthetic products from its zooxanthellae. The amount of photosynthetic carbon translocated to the coral host is often sufficient to meet its metabolic respiratory requirements. Corals may also take up dissolved organic compounds directly from seawater. Although dissolved concentrations of nutrients are very low in most tropical waters, mass transport of nutrients via diffusion or transport across coral surfaces may be sufficient, when assimilated, to supply the nutrient requirements of the algae and its host (Hoegh-Guldberg and Williamson, 1999). Corals are well adopted to this type of feeding by their extremely high surface area to volume ratio and the presence of cilia on their epidermal cells. The additional adaptation to low nutrient condition is the ability to conserve nutrients in the algae/coral unit. Animal metabolic waste products derived from holozoic feeding are retained within the coral, as they are a source of the nutrient elements (e.g. N, P) required by the zooxanthellae.

There is a general perception in the scientific community that excess of nutrients (N and/or P) in the ambient wares is one of the primary causes of degradation of stony corals and their habitats. However, after the scientific literature review on this subject, nutrients impacts on reef-building coral species appear to be much more complex.

Acropora spp. are somehow capable of avoiding internal competition for inorganic carbon, thereby maintaining high calcification rates even under conditions of elevated nutrients and increased symbiotic dinoflagellate densities (Bucher and Harrison 2002). Given the importance of Acropora species in many coral, differences between Acropora spp. and other corals were also evident in the response to elevated phosphate during ENCORE. Phosphate generally had little effect or increased calcification in *Acropora longicyathus* , *Acropora aspera* (Bucher 2000) and *Acropora palifera* (Steven and Broadbent 1997), whereas pocilloporid species in the ENCORE patch reefs showed no response or had reduced calcification in the presence of elevated phosphate.

Phosphate contamination can negatively affect corals, modifying growth rates, skeletal density, reproduction, mortality, and zooxanthellae (Dunn et al. 2012). Researchers determined the effects of elevated phosphate on coral growth and density. Genetically distinct colonies of

Acropora muricata were sub-divided and distributed among three 110-L aquaria, and exposed to phosphate levels of 0.09, 0.20, and 0.50 mg /L for four months. Total skeletal length, living tissue length, weight, branch production, and polyp extension were measured. Linear extension and tissue growth increased under all conditions. Growth rates were highest at a phosphate concentration of 0.50 mg /L. Weight increased through time, graded from low to high with phosphate concentration. Density decreased through time, and was significantly lowest in the high phosphate treatment. Phosphate concentration produced no visible effects of stress on the corals, as indicated by polyp extension and lack of mortality. It is suggested that the phosphate enhanced growth was due to increased zooxanthellar populations and photosynthetic production within the coral. Skeletal density reduction may be due to phosphate binding at the calcifying surface and the creation of a porous and structurally weaker calcium carbonate/calcium phosphate skeleton. Increased phosphate concentrations, often characteristic of eutrophic conditions, caused increased coral growth but also a more brittle skeleton. The latter is likely more susceptible to breakage and damage from other destructive forces (*e.g.*, bioerosion).

Shantz and Burkepile (2014) summarized the general patterns of N and P impacts on the growth and photobiology of reef-building stony corals. Researchers pointed out that nutrients can alter this type of coral community structures by disrupting the symbiotic relationship between nutrient-sharing zooxanthellae and oligothrophic corals. Because of such a high degree of their dependence on algae, reef-building corals are expected to be particularly vulnerable to excess of nutrients. Researchers pointed out that over a wide range of N and P concentrations, N reduced coral calcification 11%, on average, but enhanced photosynthetic rate. On the other hand, P enrichment was shown to increase average calcification rates by 9%, however, at the same time, minimally impacting photosynthetic rates. In addition, Shantz and Burkepile also reported few synergistic impacts of combined N and P on corals. Overall, the responses of corals to increasing nutrient availability were dependent on context, varied with different coral taxa and their morphology, sources of nutrient enrichment, as well as type of nutrient(s). For example, naturally occurring enrichment from fish excretion increased coral growth, while human-mediated enrichment tended to decrease coral growth.

Koop et al (2001) reported the results of 2-years experiment, which studied the biological responses of coral reefs to controlled additions of dissolved inorganic nitrogen (N) and/or phosphorus (P) on an offshore reef (One Tree Island) at the southern end of the Great Barrier Reef, Australia. In this experiment, corals were exposed to 3 different nutrient stressors: nitrogen, phosphorous and both, via lower and higher nutrient dosages. During the initial, low-dosage part of the experiment, phosphate pulses (mean dose of 2.3 μM = 0.218 mg/L) rapidly declined, reaching near-background levels (mean of 0.5 μM = 0.0475 mg/L) within 2-3 hours. As a result, fewer successfully developed embryos were formed *in Acropora longicyathus* and *A. aspera*. In *A. longicyathus* fertilization rates and lipid levels decreased. In the second, high-dosage, phase (mean dose of 5.1 μM = 0.484 mg/L, declined to mean of 2.4 μM = 0.228 mg/L), a variety of significant biotic responses occurred. Corals containing zooxanthellae assimilated phosphate rapidly and were responsive to added nutrients. Coral mortality, not detected during the initial low-dosage phase, became evident with increased nutrient dosage, particularly in *Pocillopora damicornis*. Nitrogen additions stunted coral growth, while phosphate additions had a variable effect. Coral calcification rate and linear extension increased in the presence of added phosphate but skeletal density was reduced, making corals more susceptible to breakage. Settlement of larvae from brooded species was also enhanced in phosphate treatments.

Overall, this experiment showed that reef organisms and their biological processes were impacted by elevated nutrients. Impacts were dependent on dose level, whether nitrogen and/or phosphorus were elevated and were often species-specific. The impacts were generally sub-lethal and subtle and the treated reefs at the end of the experiment were visually similar to control reefs. It is worth to point out that in this study, the phosphate level of 0.0474 mg/L was considered to be near-background level. Taking into consideration that the phosphate form of the nutrient is a component of the total phosphorus form of the numeric criterion of 0.050 mg/L for total phosphorus, appears to represent very low, near-background concentrations already present naturally in the marine ecosystem.

Parker and D'Elia (2007) provided some evidence suggesting that the elevated phosphate levels (exceeding about 1 µM [0.095 mg/L]) reduce calcification. However, firm evidence was lacking that even these levels that would be considered extremely high for reef waters, directly affect the survival of the symbiosis between algae and corals.

Wiedenmann et al. (2013) suggested that the most severe impact on coral health might not actually arise from the over-enrichment with one group of nutrients but from the resulting relative depletion of other groups. Specifically, the Wiedenmann's research considered three different combinations of nutrients: (1) a low treatment, which included a combination of approximately 0.07 µM dissolved inorganic nitrogen (DIN), represented mostly as nitrite and nitrate and approximately 0.006 µM (0.00057 mg/L) phosphate; (2) a full treatment (with 6.5 µM DIN and 0.3 µM (0.028 mg/L) phosphate, and (3) an imbalanced treatment (with 3 µM DIN and 0.07 µM (0.0066 mg/L) phosphate). Overall, this research was able to show that DIN in combination with limited phosphate concentrations resulted in an increased susceptibility of corals to temperature and light-induced bleaching. In addition, researchers pointed out that reaching favorable nutrient ratios is important in reef waters while working towards overall lower nutrient loadings, when considering changes to agricultural and waste water treatment practices.

Lastly, for the purpose of this BE, in addition to the information generated by various researchers in laboratory studies and field observations summarized above, EPA would also like to consider the recommended optimal phosphate concentrations for aquarium cultivation ranging from 0.02 to 0.05 mg/L, recommended for growth of Atlantic staghorn coral *Acropora cervicornis* (O'Neil 2015). In general, her research focused on aquarium cultivation of *A. cervicornis*, to produce fragments for reef restoration projects. The overall goal for her research over the last four years was to evaluate and compare the growth and survival of corals raised in land-based nurseries versus ocean-based nurseries, and carefully monitoring the health of aquarium-raised corals that were placed back onto a reef in Fort Lauderdale, FL. EPA believes that this information provides the additional point of view in the process of evaluation the protectiveness of existing TP criterion adopted by the USVI.

According to the scientific literature review, reef-building corals are clearly subject to many stressors, including excessive nutrients. The USVI adopted the water quality criterion for phosphorus to protect all of the USVI marine waters. As a result, the primary goal of the scientific literature review for the EPA was to evaluate the most optimal phosphorus concentrations for the recovery of the ESA-listed coral reefs. During the literature review process, the EPA identified a large amount of studies which evaluated the effects of various

nutrient conditions on different coral species and their habitats. The significant part of these studies, however, focused on nitrogen being used as a primary nutrient in the marine ecosystems. Although multiple publications focused on the combination of both, nitrogen and phosphorus, being a source of nutrients, there were only a limited amount of studies which investigated the impacts of various phosphorus concentrations (as the only nutrient) on coral reef ecosystems. In addition, the majority of studies evaluating the effects of various phosphorus concentrations in the ambient waters, investigated a phosphate as the primary form of nutrient, which made the evaluation of the total phosphorus criteria adopted by the USVI even more challenging.

In summary, as the result of the literature review, the EPA did not locate any scientific information directly related to the impacts of phosphorus on the specific ESA-listed corals of interest, except primarily Acropora genus. The Acropora genus represents over 149 stony coral species, which include two of our ESA-listed species of concern: *A. palmata* (Elkhorn coral) and *A. cervicornis* (Staghorn coral). Coral species grouped in Acropora genus share many similar characteristics. As a result, for the purposes of this BE, the EPA considered both *A. palmata* and *A. cervicornis* species to be sensitive to phosphorus levels in the ambient waters to the same degree what Acropora species evaluated above.

Furthermore, the EPA was not able to locate any scientific information directly related to the phosphorus impacts on five newly listed coral species of concern: Pillar coral, *Dendrogyra cylindrus;* Lobed star coral, *Orbicella annularis;* Mountainous star coral, *Orbicella faveolata*; Boulder star coral, *Orbicella franksi* and Rough cactus coral, *Mycetophyllia ferox*. Because three-quarters of reef-building (stony) coral species were reported to spawn gametes and rely on external fertilization and planktonic development, Albright et al. (2010) considered elkhorn coral *A. palmata* to be a representative of spawning species in general. The EPA agrees with this approach, and after detailed evaluation of characteristics for all three Genus: Dendrogyra, Orbicella and Mycetophyllia and their anatomic as well as habitat similarities, the EPA did not come across any information precluding the Agency from the assumption that these five coral species will be protected by the phosphorus criterion adopted by the USVI to the same degree as two Acropora corals.

The final listing rule (NMFS 2006) identified nutrients as a threat contributing to the threatened status of elkhorn and staghorn corals. Likewise, the final rule maintaining elkhorn and staghorn coral as threatened species and introducing the remaining coral species to the ESA corals (NMFS 2014), lists nutrient over-enrichment as a threat contributing to the status of the species. It is widely understood that excess nutrients on coral reefs can lead to algal overgrowth and competition if levels of herbivory are inadequate to remove excess algal production. However, nutrient effects on corals (severity, magnitude, and source) are complex and highly debated. Furthermore, the effects of nutrient loads on coral physiology are currently unknown, relative to other stressors for same coral species. For this reason, while nutrients are recognized as a threat likely to impede the recovery of these corals, the ranking of "significant but unknown" was provided to listed corals for all regions.

Nutrients are known to negatively impact corals. However, there is a lack of information tying presence of nutrients on reefs to coral condition and a lack of information regarding threshold of tolerance to this threat. Once baseline information on levels of nutrients, in robust reference populations, has been determined, a measurable criterion can be developed

Conclusion: During the process of the literature review, the EPA did not come across any research recommending the most optimal phosphorus concentrations for the recovery of any of seven ESA-listed coral species. Based on the limited research summarized above, the EPA has determined that the numeric nutrient criterion adopted by the USVI for Total Phosphorus of 0.050 mg/L is NLAA seven ESA-listed coral species.

The EPA is in the process of assisting the VIDPNR with the derivation of numeric nutrient criteria for total nitrogen to protect sensitive coral reef species. The EPA plans to initiate conversations with NOAA-NMFS, as we get closer to criteria derivation. In addition, the EPA plans to work closely with the VIDPNR and NOAA-NMFS to reevaluate the existing criterion for total phosphorus and revise it, if necessary to ensure that all of the ESA-listed coral species are being protected. The potential revisions to the numeric nutrient criteria will be considered during the next triennial WQSR review scheduled to be completed by 2018.

b) Analysis of potential impacts of phosphorus on Whales

- Blue whale, *Balaenoptera musculus;* Fin whale, *Balaenoptera physalus;* Sei whale, *Balaenoptera borealis;* Sperm whale, *Physeter macrocephalus* and Humpback whale, *Megaptera novaeangliae.*

During the scientific literature review, the EPA did not come across any research which reported on the specific effects of various phosphorus concentrations on ESA-listed whale species. In addition, the EPA did not find any information which suggested that the total phosphorus concentration adopted by the USVI as water quality criterion would pose a direct threat to the recovery or survival of any of five ESA-listed species or their habitats.

Conclusion: During the process of the literature review, the EPA did not come across any research recommending the most optimal phosphorus concentrations for the recovery of any of five ESA-listed whale species. The EPA has determined that the numeric nutrient criterion adopted by the USVI for Total Phosphorus of 0.050 mg/L is NLAA ESA-listed whale species or their habitats.

The EPA is in the process of assisting the VIDPNR with the derivation of numeric nutrient criteria for total nitrogen and reevaluation of the existing criterion for total phosphorus. The potential revisions to the numeric nutrient criteria will be considered during the next triennial WQSR review scheduled to be completed by 2018.

c) Analysis of potential impacts of phosphors on Sea Turtles

- Hawksbill sea turtle, *Eretmochelys imbricata*; Green sea turtle, *Chelonia mydas*;
- Loggerhead sea turtle, *Caretta caretta*; Leatherback sea turtle, *Dermochelys coriace*

During the scientific literature review, the EPA did not come across any research which would report on the specific effects of various phosphorus concentrations on ESA-listed sea turtle species. In addition, the EPA did not find any information which would suggest that the total phosphorus concentration adopted by the USVI as water quality criterion would pose a direct threat to the recovery or survival of any of four ESA-listed species or their habitats.

Conclusion: During the process of the literature review, the EPA did not come across any research recommending the most optimal phosphorus concentrations for the recovery of any of four ESA-listed turtle species. The EPA has determined that the numeric nutrient criterion adopted by the USVI for Total Phosphorus of 0.050 mg/L is NLAA ESA-listed sea turtle species.

The EPA is in the process of assisting the VIDPNR with the derivation of numeric nutrient criteria for total nitrogen to protect sensitive coral reef species. The EPA plans to initiate conversations with NOAA-NMFS, as we get closer to criteria derivation. In addition, the EPA plans to work closely with the VIDPNR and NOAA-NMFS to reevaluate the existing criterion for total phosphorus and revise it, if necessary to ensure that all of the ESA-listed coral species are being protected. The potential revisions to the numeric nutrient criteria will be considered during the next triennial WQSR review scheduled to be completed by 2018.

d) Analysis of potential impacts of phosphorus on Fish

- Scalloped Hammerhead Shark, *Sphyrna lewini* and Nassau Grouper, *Epinephelus striatus*

During the scientific literature review, the EPA did not come across any research which would report on the specific effects of various phosphorus concentrations on ESA-listed fish species. In addition, the EPA did not find any information which would suggest that the total phosphorus concentration adopted by the USVI as water quality criterion would pose a direct threat to the recovery or survival of any of ESA-listed species or their habitats.

Conclusion: During the process of the literature review, the EPA did not come across any research recommending the most optimal phosphorus concentrations for the recovery of any of ESA-listed fish species. The EPA has determined that the numeric nutrient criterion adopted by the USVI for Total Phosphorus of 0.050 mg/L is NLAA ESA-listed fish species.

The EPA is in the process of assisting the VIDPNR with the derivation of numeric nutrient criteria for total nitrogen to protect sensitive coral reef species. The EPA plans to initiate conversations with NOAA-NMFS, as we get closer to criteria derivation. In addition, the EPA plans to work closely with the VIDPNR and NOAA-NMFS to reevaluate the existing criterion for total phosphorus and revise it, if necessary to ensure that all of the ESA-listed coral species are being protected. The potential revisions to the numeric nutrient criteria will be considered during the next triennial WQSR review scheduled to be completed by 2018.

7. Toxic Pollutants - Organic Compounds

a) Analysis of potential impacts of organic compounds on Corals

(1) Pesticides/Herbicides

There is a significant amount of information published in the scientific literature which relates to the overall impacts of pesticide/herbicides on the marine environment, including coral reefs. There is a general understanding among scientists that coral reefs ecosystems are being negatively impacted by increased concentrations of pesticides present in the water. In general, at low concentrations, pesticides have been shown to reduce the photosynthesis of coral symbionts (the dinoflagellates *Symbiodinium* spp.), while at higher concentrations pesticides can damage the partnership between coral and symbiont,(algae) resulting in loss of symbionts from the coral host, causing bleaching (Jones and Kerswell 2003; Negri et al. 2005). Studies evaluating the effects of the specific pesticides (at specific concentrations) on corals are very limited. Such information, as it relates to the ESA-listed species of corals was not identified by the EPA. The EPA assembled the most relevant information as basis for determinations and the summary are presented below.

(a) Carbaryl

Acevedo (1991). investigated effects of carbaryl on planulae of the hermatypic coral *Pocillopora damicornis*. Planulae larvae of *P. damicornis* were exposed carbaryl in concentrations of 0.01, 0.1, 1, 10, and 100 mg/L. Actively swimming planulae were held in test solutions for 96 hr., after which viability was determined. Carbaryl in concentrations up to 10 mg/L had no effect on the planulae after 96 hours, causing larval mortality only at high concentrations. Concentrations of 100 µg/L killed 70 to 90% of planulae in all three replicates within the first 12 hours of the experiment. All of the evaluated in this study concentrations of carbaryl were above the 1.6 µg/L criterion adopted by the USVI.

Markey et. al., (2007) compared the sensitivity of coral *Acropora millepora* gametes, larvae and adult branches exposed to carbaryl in a series of laboratory experiments to determine the threshold concentrations at which carbaryl became toxic to key life-history events and stages. In this study, researchers investigated four replicate vials with concentrations ranging from 0.3 to 30 µg /L of carbaryl. Carbaryl did not inhibit coral fertilization within the concentration range used in this study. Six day-old larvae were exposed to 30, 100 and 300 µg /L of carbaryl. Metamorphosis was completely inhibited at these concentrations. The 7 and 8 day old larvae were exposed to lower carbaryl concentrations (1.0, 3.0, 10 and 30 µg /L for 7 day and 0.1, 0.3 and 1.0 µg /L for 8 day larvae). Larval settlement and metamorphosis were reduced by 50 to 100% following 18 hour- exposures to carbaryl treatments. Adult branches of *A. millepora* were exposed to carbaryl in 3 successive 96 hour experiments. Bleaching was observed on the tips of many branches in 10 µg /L contaminant treatments. The overall, researchers reported the LOEC for carbaryl to be 3.0 µg /L, which is significantly above the water quality criterion adopted by the USVI.

Armbrust and Crosby (1991) studied the fate of carbaryl in filter-sterilized and raw (unfiltered) seawater. Researchers reported that carbaryl in the dark was hydrolyzed to l-naphthol with a half-life of 24 hours at pH 7.9 or 23 hours at pH 8.2 (24°C). Naphthol was degraded to undetectable levels in 96 hours in raw seawater. In sunlight, carbaryl degraded with a half-life of 5 hours and l-naphthol was completely degraded after 2 hours. No further degradation products were observed for either compound. These data suggested that carbaryl may not be stable enough in the ambient seawater to permit exposure of susceptible marine life, while, in the presence of sunlight, carbaryl would rapidly dissipate to undetectable levels.

Conclusion: During the scientific literature review, the EPA did not come across any research which would evaluate the effects of various carbaryl concentrations on ESA-listed coral species, directly. Due to a very limited amount of research published on this subject, the EPA evaluated effects of carbaryl on other coral reefs species. The EPA did not find any information which would suggest that the carbaryl concentration of 1.6 µg/L would pose a threat to the recovery of any of the ESA-listed coral species. Based on the limited research summarized above, the EPA has determined that the water quality acute criterion of 1.6 µg /L, adopted by the USVI for carbaryl is NLAA seven ESA-listed coral species or their habitats.

Should additional information related to the effects of carbaryl on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(b) Chlordane

Various studies indicate that the chlordane can accumulate in coral tissue (Olafson 1978), however, its effects on corals are not yet fully understood. Firman and Gassman (1995) examined the effects of long-term sublethal chlordane exposure to adult colonies and larvae of the scleractinian coral, *Montastraea faveolata*. The acute toxicity of chlordane to *M. faveolata*, another scleractinian coral, *Porites divaricata*, and a zooxanthellate tropical anemone were also examined. Corals were very tolerant of chlordane in short term tests with 96 hour LC50 for *P. divaricata* of 15.3 µg/L and 96 hour LC50 for *M. faveolata* of 17.8 µg/L. A 90-day exposure to 1 to 15 µg/L chlordane was also investigated by researchers. A 90-day exposure to 10 µg/L chlordane depressed photosynthesis, respiration, caused bleaching by impacting algal densities and chlorophyll content, caused death for 50% of the exposed corals. Exposure to 10 µg/L also caused developmental abnormalities in larvae of *M. faveolata.*

Corals exposed for 90 days to 1 µg/L chlordane attained normal rates of photosynthesis after 2 weeks in clean seawater; respiration rates returned to normal levels, but chlorophyll concentrations stayed significantly depressed after 5 weeks of recovery. All of the evaluated in this study concentrations of chlordane were above concentrations adopted by the USVI as water quality criteria.

Conclusion: During the scientific literature review, the EPA did not find any information which would suggest that the chlordane acute concentrations of 0.09 µg/L and chronic concentration of 0.004 µg/L would pose a threat to the recovery of any of the ESA-listed coral species. The EPA

has determined that the above values adopted by the USVI for chlordane are NLAA seven ESA-listed coral species and their habitats.

Should additional information related to the effects of chlordane on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(c) Chlorpyrifos

Acevedo (1991) investigated effects of chlorpyrifosl on planulae of the hermatypic coral *Pocillopora damicornis*. Planulae larvae of the reef coral *P. damicornis* were exposed to chlorpyrifos in concentrations ranging from 0.01 to 100 mg/L. Actively swimming planulae were held in test solutions for 96 hours, after which their viability was determined. Chlorpyrifos in concentrations of 0.01 mg/L had no noticeable adverse effects on the planulae. Concentrations of 0.1 mg/L slowed the swimming motion considerably in all three replicates, while mortality was observed only at higher concentrations. Concentrations of 1 mg/L killed 50 to 100% of the planulae within 24 hours, accompanied by a decrease in the speed of motion for the ones that were still alive. Concentrations of 10 mg/L killed all planulae within the first 12 hours of experimentation.

In contrast, adult coral branches were reported to be more sensitive to chlorpyrifos. Te (1998) investigated the effect of a chlorpyrifos based pesticide on a reef-building coral *Pocillopora damicornis*. Short-term (96 hours) static bioassays with renewal of toxicant every 24 hours were conducted using chlorpyrifos concentration of 6 μg /L. The bioassay tests revealed high sensitivity of the coral *P. damicornis* to toxicant, with 50% of individuals dying at 6 μg /L (LC50 = 6 μg /L).

Markey et al. (2007) compared the sensitivity of gametes, larvae and adult branches of the broadcast-spawning coral *Acropora millepora,* exposed to chlorpyrifos in a series of laboratory experiments to determine the threshold concentrations at which it became toxic to key life-history events and stages. In this study, researchers investigated chlorpyrifos using four replicate vials with treatment concentrations ranging from 0.3 to 30 μg /L.

Chlorpyrifos did not inhibit coral fertilization within the concentration range used in this study. Six day-old larvae were exposed to 30, 100 and 300 μg /L of chlorpyrifos. Metamorphosis was completely inhibited at these concentrations. The 7 and 8 day old larvae were exposed to lower chlorpyrifos concentrations (1.0, 3.0, 10 and 30 μg /L for 7 day and 0.1, 0.3 and 1.0 μg /L for 8 day larvae). Settlement and metamorphosis were reduced by 50 to 100% following 18 hours-exposure to very low concentrations (0.3 to 1.0 μg l/L) of chlorpyrifos. Adult branches of *A. millepora* were exposed to toxicant in 3 successive 96 hours experiments. Paling (apparent bleaching) was observed visually on the tips of many branches in 10 μg /L contaminant treatments. The overall, researchers reported the LOEC for chlorpyrifos to be 1.0 μg /L, which is significantly above the water quality criterion adopted by the USVI.

Conclusion: During the scientific literature review, the EPA did not find any information which would suggest that the chlorpyrifos acute concentrations of 0.011 μg/L and chronic concentration of 0.0056 μg/L would pose a threat to the recovery of any of the ESA-listed coral species. The EPA has determined that the above values adopted by the USVI for chlordane are NLAA seven ESA-listed coral species and their habitats.

Should additional information related to the effects of chlorpyrifos on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(d) Dieldrin

McCloskey and Chesher (1971) subjected three different species of corals (*Orbicella annularis, Acropora cervicornis,* and *Madracis mirabilis*) to 1,000 μg/L mixture of p, p'-DDT, dieldrin, and Aroclor 1254 (a PCB) in equal proportions. Colonies were also exposed to 10, 100, and 1,000 μg/L of each of the three compounds. Scientists reported no observed changes in coral's feeding behavior, polyp extension, sediment clearing or settling. However, the authors recorded an increase in respiration and a decrease in photosynthesis for all 3coral species, such that the ratio of respiration to photosynthesis fell below 1.0. Photosynthesis remained depressed for up to 4 days, at which time the experiment was terminated. The compensation value (i.e. the light level at which oxygen generated by photosynthesis is equal to oxygen consumed by respiration) also increased with the addition of the organochlorides. The authors concluded that the mixture affected coral metabolism such that the growth and maintenance of the coral colonies could have been compromised. All concentrations investigated in this study, however, were significantly above the levels adopted for dieldrin by the USVI as water quality standards.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of dieldrin on coral ecosystems. In addition, the EPA did not find any information which would suggest that the dieldrin concentrations at or below 0.71 μg/L would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criterion adopted by the USVI for dieldrin of 0.71 μg/L (as acute value) and 0.0019 μg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of dieldrin on coral species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(e) Endosulfan (Alpha- and beta)

Markey et. al. (2007) compared the sensitivity of gametes, larvae and adult branches of the broadcast-spawning coral *Acropora millepora* exposed to endosulfan in a series of laboratory experiments to determine the threshold concentrations at which it became toxic to key life-

history events and stages. In this study, researchers investigated this toxicant in treatment concentrations ranging from 0.3 to 30 µg /L. Endosulfan did not inhibit coral fertilization with concentrations up to 30 µg/L. The 7 and 8 day old larvae were exposed to lower endosulfan concentrations (1.0, 3.0, 10 and 30 µg /L for 7 day and 0.1, 0.3 and 1.0 µg /L for 8 day larvae). Larval settlement and metamorphosis were reduced by 50 to 100% following 18 hour- exposures to very low concentrations (0.3 to 1.0 µg l/L) of endosulfan. Adult branches of *A. millepora* were exposed to endosulfan in 3 successive 96 hour experiments. Bleaching was observed on the tips of many branches in 10 µg /L contaminant treatments. The overall, researchers reported the LOEC for endosulfan to be 1.0 µg /L, which is significantly above the water quality criterion adopted by the USVI.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of endosulfan on coral ecosystems. In addition, the EPA did not find any information which would suggest that the endosulfan concentrations at or below 0.034 µg/L in ambient water would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criterion adopted by the USVI for endosulfan (both, alpha and gamma) of 0.034 µg/L (as acute value) and 0.0087 µg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of endosulfan on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(f) 4, 4'DDT

McCloskey and Chesher (1971) subjected the different species of corals (*Orbicella annularis, Acropora cervicornis,* and *Madracis mirabilis*) to 1,000 µg/L mixture of p, p'-DDT, dieldrin, and Aroclor 1254 (a PCB) in equal proportions. Colonies were also exposed to 10, 100, and 1,000 µg/L of each of the three compounds. Scientists reported no changes in feeding behavior, polyp extension, sediment clearing or settling of coral associates were observed. However, the authors recorded an increase in respiration and a decrease in photosynthesis for all 3 species, such that the ratio of respiration to photosynthesis fell below 1.0. Photosynthesis remained depressed for up to 4 days, at which time the experiment was terminated. The compensation value (i.e. the light level at which oxygen generated by photosynthesis is equal to oxygen consumed by respiration) also increased with the addition of the organochlorides. The authors concluded that the organochloride mixture affected coral metabolism such that the growth and maintenance of the coral colonies could have been compromised. All concentrations investigated in this study, however, were significantly above the levels adopted for 4, 4 DDT by the USVI as water quality standards.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of 4, 4 DDT on coral ecosystems. In addition, the EPA did not find any information which would suggest that concentrations of 4,4DDT at 0.13 µg/L or below would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for 4,4 DDT

of 0.13 μg/L (as acute value) and 0.001 μg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of 4, 4 DDT on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(g) Other pesticides/herbicides

In this subsection, the EPA identified the list of pesticides for which the USVI adopted EPA-recommended water quality criteria, but for which there was no information identified in the literature search related to their effects on coral ecosystems. The pesticides listed below were subject of investigations related to their accumulation in coral tissue and sediments. However, the EPA did not come across any research which would report on their specific effects on corals, at their specific concentrations indicating potential "protective" thresholds. In addition, the EPA did not find any information which would suggest that concentrations adopted as water quality criteria would pose a threat to the recovery of any of the ESA-listed coral species.

As a result, the EPA has determined that the water quality criteria adopted by the USVI for the following pesticides are NLAA the recovery of seven ESA-listed coral species or their habitats: toxaphene, pentachlorophenol, mirex, methoxychlor, malathion, heptachlor and heptachor epoxide, guthion, lindane, endrin, diazinon, demeton and aldrin.

As soon as the additional information related to the toxicity of the above listed pesticides on coral species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of the sensitive coral reef ecosystems.

(2) Nonylphenol

Nonylphenol is an organic compound used in manufacturing of lubricating oil additives, laundry and dish detergents, emulsifiers, and solubilizers (Soares et al. 2008). Nonylphenol persists in aquatic environments and is moderately bioaccumulative. It is not readily biodegradable, and it can take months or longer to degrade in surface waters. Due to this bioaccumulation and persistence of nonylphenol, it has been suggested that nonylphenol could be transported over long distances and have a global reach that stretches far from the site of contamination.

It is well established in the scientific literature that nonylphenol present in plastic bags can leach from the bag into seawater at such high concentrations that it can be deadly to coral reef fish (Hamlin et al. 2015). However, its direct impacts on coral species are not yet known.

Shafir et al. (2014) examined ecotoxicological impacts (survivorship, growth) of nonylphenol ethoxylate (NPE) on two branching coral species (*Stylophora pistillata* and *Pocillopora*

damicornis). Nubbins assays, after 24-hours exposure to NPE were monitored for 203 days and revealed high mortality in 1 and 5 mg/L NPE concentrations. Assays further showed species-specific mortalities with *Stylophora* LC50 of 3.03 mg/L and *Pocillopora* LC50 of 2.26 mg/L.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of nonylphenol on ESA-listed coral species of interest. In addition, the EPA did not find any information which would suggest that concentrations of nonylphenol at or below 7 μg/L would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for nonylphenol of 7 μg/L (as acute value) and 1.7 μg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of nonylphenol on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(3) Tributyltin (TBT)

Tributyltin (TBT) is an umbrella term for a class of organotin compounds (tin [Sn] with hydrocarbon bond). For the last four decades TBT was used as a biocide in anti-fouling paint, which was applied to the hulls of ocean going vessels. With time, the TBT was found to slowly leach out into the marine environment where it is highly toxic to a wide range of organisms at the bottom of the food chain. The TBT is likely to biomagnify up the marine predators' food web, harmfully affecting invertebrates and vertebrates.

Negri and Heyward (2001) evaluated inhibition of fertilization and larval metamorphosis of the coral *Acropora millepora* in response to solution of the TBT using laboratory-based bioassays. Nominal concentrations that inhibited 50% fertilization and metamorphosis (IC50) were calculated from 4-hours fertilization and 24-hours metamorphosis assays and were based on introduced dose. TBT inhibited fertilization in 50% of *A. millepora* gametes relative to control assays at concentration of 200.31 μg/L (IC50) and inhibited metamorphosis in 50% of *A. millepora* larvae at concentration of 2.0 μg/L.

TBT was shown to inhibit the synthesis of proteins and their subsequent incorporation into skeleton of the coral Stylophora pistillata (Allemand et al. 1998). Researchers reported that TBT significantly inhibited protein synthesis and the subsequent incorporation of protein into coral skeleton. This effect was correlated with a reduction in the rate of calcification. Protein synthesis was shown to be the parameter most sensitive to TBT, with IC50 of 0.2 uM/L).

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of TBT on ESA-listed corals. In addition, the EPA did not find any information which would suggest that concentrations of TBT at 0.42 μg/L or below would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the

EPA has determined that the water quality criteria adopted by the USVI for tritutyltin of 0.42 µg/L (as acute value) and 0.0074 µg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of TBT on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(4) Polychlorinated biphenyls (PCBs)

Polychlorinated biphenyls (PCBs) are a group of widespread contaminants, and accumulation of PCBs has been observed in corals in the field. However, the toxic effects of PCBs on corals have not been investigated.

Chen et al. (2012) tested short and long term toxicity of Aroclor 1254, a commercial PCB mixture, on the scleractinian coral *Stylophora pistillata*. Coral nubbins were incubated in either control seawater or seawater dosed with PCBs (approximately 0.3µg/L) for 96 hours. After the exposure, coral nubbins were moved into clean seawater and their survival and growth were observed for another 50 days. All nubbins survived during the exposure and the following 50-day recovery period. In conclusion, acute exposure of *S. pistillata* to Aroclor 1254 at 0.3 µg/L did not affect coral survival, photosynthesis or growth.

McCloskey and Chesher (1971) subjected the different species of corals (*Orbicella annularis, Acropora cervicornis,* and *Madracis mirabilis*) to 1,000 µg/L mixture of p, p'-DDT, dieldrin, and Aroclor 1254 (a PCB) in equal proportions. Colonies were also exposed to 10, 100, and 1,000 µg/L of each of the three compounds. Scientists reported no changes in feeding behavior, polyp extension, and sediment clearing or settling of coral associates were observed. However, the authors recorded an increase in respiration and a decrease in photosynthesis for all 3 species, such that the ratio of respiration to photosynthesis fell below 1.0. Photosynthesis remained depressed for up to 4 days, at which time the experiment was terminated. The compensation value (i.e. the light level at which oxygen generated by photosynthesis is equal to oxygen consumed by respiration) also increased with the addition of the organochlorides. The authors concluded that the organochloride mixture affected coral metabolism such that the growth and maintenance of the coral colonies could have been compromised.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of PCBs on ESA-listed corals. In addition, the EPA did not find any information which would suggest that concentrations of PCBs of 0.03 µg/L would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criterion adopted by the USVI for PCBs of 0.03 µg/L (as chronic value) is NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of PCBs on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(5) Analysis of potential impacts of organic compounds on Whales

Some of the most deadly and widespread of pollutants are human-made chemicals containing carbon and chlorine, called organochlorines. Such chemicals are used as pesticides and herbicides, in the dry cleaning industry, in tanneries, and in electrical equipment. Organochlorines are lipophilic, thus they accumulate in fats and oil rather than water. Therefore, they tend to build up in the fatty tissues of marine organisms such as whales. Usual biological processes do not break them down, which makes them very persistent in the environment and easy to bioaccumulate up the food chain (with increasing in concentration with each level). In the oceans, organochlorines reach their highest levels in the top predators, which are sharks and marine mammals such as whales.

Organochlorines in whales have the potential to disrupt a variety of biochemical and physiological processes. Organochlorines concentrate in whales' large blubber reserves. At times, when they are migrating, whales will not feed and so will live off these reserves. If the reserves are contaminated with pollutants it may cause the animal to become ill. In addition to this, organochlorines are transferred from mother to calf during lactation, or nursing.

Marine mammals accumulate high concentrations of persistent organic contaminants from the invertebrates and fish they consume because they have high lipid contents, and they may also metabolize these compounds more slowly than terrestrial mammals. Body burdens of organochlorine pollution are higher in the toothed whales (such as Sperm whale) than the baleen whales because they feed higher up the food chain. Organochlorines severely inhibit the reproductive processes of whales and can lead to infertility, birth defects, tumors and lesions, tooth decay, adrenal gland cysts, a high %age of malignant growths, and carcinogens.

- Blue whale, *Balaenoptera musculus*

According to NOAA (1998), the planktivorous diet of Blue whales makes them less susceptible than piscivorous (fish-based diet) baleen whales to the accumulation of organochlorine and metal contaminants in their tissue. O'Shea and Brownell (1994) reported that there is no reason to suspect that levels of pollutants in any baleen whales are presently high enough to cause toxic or other effects.

Metcalfe et al. (2004) investigated the differences in contaminants concentration in blubber of female and male blue whales. Among organochlorine compounds, concentrations of heptachlor and aldrin were generally below detection limits in samples obtained from a female, although these compounds were detected in a few samples obtained from a male. However, heptachlor epoxide was present in all biopsy samples. Concentrations of endrin, endrin aldehyde,

methoxychlor and g-HCH were in general very low or below detection limits. The classes of compounds present in the highest concentrations in both female and male were DDT and its metabolites, PCBs, HCHs, chlordanes, HCB and mirex. Statistical analysis showed that concentrations of total DDT and total PCB were significantly lower in the blubber of females relative to males. Although there was a similar trend for HCB and total chlordanes, the differences between sexes were not statistically significant. No sex-related differences were observed for concentrations of HCH compounds, mirex, heptachlor epoxide, endrin aldehyde and methoxychlor. The lower concentration of contaminants in females (compared to males) most likely reflects the loss of these hydrophobic compounds through maternal transfer to young. This indicates that calves of blue whales may be exposed to PCBs and organochlorines as a result of maternal transfer through transplacental routes and lactation, which is further consistent with the fact that concentrations of organic contaminants in the calf were found to be within the same range as the mean concentrations found in the female blue whales.

Among blue whales, there were differences in the concentrations of PCBs and DDT compounds in the blubber of males and females, but no differences were observed in the patterns of these compounds. In Metcalfe et al. study, there were sex-related differences in the concentrations of PCBs and DDT compounds, but this trend was not observed for several other organochlorine contaminants, such as HCB, HCHs, and the metabolites of hexachlorocyclodiene pesticides (e.g. endrin aldehyde, heptachlor epoxide). It appears that several less persistent compounds do not show the same high degree of loss from female whales as observed for more persistent PCBs and DDT compounds. The data from Metcalfe et al. study indicate that PCB congeners and organochlorine compounds are present in the blubber of whale calves in concentrations and proportions that are similar to the blubber of the females. This trend has significance for all cetaceans (whales or dolphins) in which concentrations of contaminants are very high, the sensitive early life stages of these animals may be exposed to toxicologically significant concentrations of contaminants.

In general, it appears that the pattern of PCB congeners is the same in the blubber of both males and females. Similarly, the relative proportions (%) of DDT and its metabolites are consistent among both sexes of blue whale. The data on the proportions of HCH isomers and chlordane compounds in male and female blue whales show similar trends.

In studies of male and female of the blue whale, Metcalfe et al. reported concentrations of 25 PCB congeners, DDT and metabolites and several other organochlorine compounds to be present at higher concentrations in males relative to females; reflecting maternal transfer of these persistent contaminants from females into young (Metcalfe et al 2004). Sex-related differences in concentrations were not observed with less persistent contaminants, such as HCHs. In humpback whale samples, there were no significant differences in the concentrations of PCBs and organochlorine compounds in the blubber of females and calves. These data indicate that calves quickly bioaccumulate contaminants by transplacental and lactational routes to concentrations that are in equilibrium with females.

Accumulated by blue whales contaminants have been also studied by Trumble and Usenko. Lifetime contaminant profiles have been reconstructed for an individual male blue whale using the earplug as a natural aging matrix that is also capable of archiving and preserving lipophilic compounds (Trumble and Usenko 2013). Many large baleen whale (family Balaenopteridae)

species are known to accumulate layers of cerumen (i.e., ear wax) from birth in the ear canal, yielding an earplug composed of lipids, waxes, and keratin, accumulating continuously, producing alternating dark- and light-colored layers, which have been shown to be associated with periods of feeding and migration. Historically, the analysis of earplugs has enabled the accurate estimation of the age of whales in a manner that is similar to counting growth rings in trees. Trumble and Usenko used a male blue whale earplug to age and reconstruct lifetime profile (i.e., birth to death) of a wide range of hormones, persistent organic pollutants (POPs), and mercury. This investigation has demonstrated that contaminants and hormones that routinely accumulate in whale blubber also accumulate in whale earplug, at quantifiable concentrations. Study further demonstrated that lipophilic compounds accumulating in cerumen are chronologically archived and their concentrations in the individual layers can be measured and used to construct lifetime profiles of exposure to different contaminants.

Trumble and Usenko reconstructed these profiles with a 6-months resolution for a wide range of analytes including contaminants such as pesticides, flame retardants and mercury. Early periods of the reconstructed contaminant profiles for pesticides (such as chlordanes and dichlorodiphenyltrichloroethanes), polychlorinated biphenyls, and polybrominated diphenyl ethers demonstrated significant maternal transfer occurred at 0–12 months. The total lifetime organic contaminant burden measured between the earplug (sum of contaminants in laminae layers) and blubber samples from the same organism were similar. The lifetime mercury profile reconstructed from the 24 layers showed different periods of peak exposure compared with the organic contaminants. Total mercury profiles revealed reduced maternal transfer and two distinct pulse events compared with organic contaminants.

- Fin whale, *Balaenoptera physalus*

Based on studies of contaminants in baleen whales, including fin whales, and other marine mammals, organic and heavy metal pollutants do not appear to be a major threat to fin whales in most areas where fin whales are found (O'Shea and Brownell, 1994). This research indicates that concentrations of organochlorine and metal contaminants in tissues of baleen whales are low, and lower than other marine mammal species. They further state that there is no firm evidence that levels of organochlorines, organotins, or heavy metals in baleen whales generally are high enough to cause toxic or other damaging effects. Weisbrod et al. (2000) studied the organochlorine exposure and bioaccumulation in North Atlantic baleen Right whale (*Eubalaena glacialis*) and reported that biopsy concentrations are an order of magnitude lower than the blubber burdens of seals and toothed whales. Study did not provide any evidence that Right whales bioaccumulated hazardous concentrations of organochlorines, which was consistent with other surveys of baleen whales (Weisbrod et al. 2000). Among baleen whales, Aguilar (1983) observed that mean levels of DDT and PCBs in the North Atlantic fin whales were significantly lower (0.74 and 12.65, respectively) than in a study of North Atlantic sperm whales (4.68 and 26.88 respectively).

In general, the threat from contaminants and pollutants occurs at a low severity and there is a medium level of uncertainty (NOAA 2011b). Thus, the relative impact to recovery of fin whales due to contaminants and pollution is ranked as low. Little is known about the possible long-term and trans-generational effects of exposure to pollutants. Aguilar and Borrell (1994) note that while pollutant burdens in young fin whale specimens from the two sexes were indistinguishable,

from the onset of sexual maturity, concentrations of all organochlorines increased with age and body size in males and decreased in females until both reached a plateau. The decrease observed in female blubber concentrations was attributed to reproductive transfer, mainly though lactation.

- Sei whale, *Balaenoptera borealis*

As stated by NOAA in the Final Recovery Plan For The Sei Whale (*Balaenoptera borealis*) (NOAA 2011a), contaminants and pollutants were determined to be a low threat for Sei whales.

Based on studies of contaminants in baleen whales, pollutants do not appear to be a major threat to sei whales in most areas where sei whales are found. O'Shea and Brownell (1994) indicated that concentrations of organochlorine contaminants in tissues of baleen whales were low, and lower than other marine mammal species. They further stated that there was no firm evidence that levels of organochlorines and organotins in baleen whales generally were high enough to cause toxic or other damaging effects. In a study of organochlorine exposure and bioaccumulation in another baleen whale, the North Atlantic right whale (*Eubalaena glacialis)*, Weisbrod et al. (2000) noted that biopsy concentrations were an order of magnitude lower than the blubber burdens of seals and odontocetes. They concluded that there was no evidence to indicate that right whales bioaccumulate hazardous concentrations of organochlorines, and further noted that these were consistent with similar studies of baleen whales (Weisbrod et al. 2000).

The highest concentrations of organochlorines found in cetaceans, including sei whales, are in the Mediterranean Sea (Aguilar et al. 2002). High concentrations of organochlorines in cetaceans also occur, although to a lesser extent, along the Pacific coast of the U.S. and generally in other mid-latitudes in the Northern Hemisphere. Sei whales in some locations are known to accumulate DDT, DDE, and PCBs (Henry and Best 1983; Borrell and Aguilar 1987; Borrell 1993). Males tend to carry larger burdens than females, as gestation and lactation transfer these toxins from mother to offspring, thereby lowering levels in mothers. However, there was no evidence reported that these or other contaminants are a threat to sei whale populations.

NOAA further indicated that the sei whale's strong preference for copepods and euphausiids (*i.e.*, low trophic level organisms), at least in the North Atlantic, may make it less susceptible to the bioaccumulation of organochlorine contaminants than, for example, fin, humpback, and minke whales, all of which seem to feed more regularly on fish and euphausiids (O'Shea and Brownell Jr. 1994). Because sei whales off California often feed on pelagic fish as well as invertebrates (Rice 1977), they might accumulate contaminants to a greater degree than do sei whales in the North Atlantic.

There is no evidence that levels of organochlorines and organotins in baleen whales generally (including sei whales) are high enough to cause toxic or other damaging effects (O'Shea and Brownell 1994). It should be emphasized, however, that very little is known about the possible long-term and trans-generational effects of exposure to pollutants, or about the possible compounding effects of exposure to two or more pollutants, in virtually any marine mammal species.

- Sperm whale, *Physeter macrocephalus*

Marsili et al. (2014) reported that in 2009, seven sperm whales were found to be stranded along the Adriatic coast of Southern Italy. Necropsies were completed on three and muscle and blubber were collected from the other four for analysis for pollutants. The whales were found to have high levels of organochlorine xenobiotics (immunosupressors) in their blubber and also high levels of polycyclic aromatic hydrocarbons (the most toxic family of hydrocarbons) in their muscle. Biomarkers as indicators of exposure to contaminants were elevated in the liver and skin, indicating that the whales were under significant toxicological stress. Although the cause of death for these stranded sperm whales was unknown, the researchers expected that the contaminant loads may have lowered their immune system defenses and made them more susceptible to disease. This theory, however was not verified.

As reported by NOAA (2010b), Holsbeek et al. (1999) analyzed tissue samples obtained from 21 sperm whales that stranded in the North Sea in 1994/1995. Results indicate that PCB, DDE, and PAH levels were low and similar to levels reported for other marine mammals. While these strandings were not attributable to contaminant burdens, Holsbeek et al. (1999) do suggest that the stable pollutants might affect the health or behavior of North Atlantic sperm whales.

Levels of organochlorine contaminants in sperm whales that stranded dead off northwestern Spain were intermediate between the levels found in fin whales (baleen whale) and small odontocetes (toothed species) in the same region (Aguilar 1983). In addition, this research indicated that the levels of contaminants bioaccumulated in females were consistently higher than those in males. This is not expected taking into consideration placental and milk transfer from mothers to their young, which usually results in a lower level of bioaccumulation in adult females compared to mails. Researchers suggest that this phenomena most likely is a result of males feeding in less polluted waters on less contaminated prey.

- Humpback whale, *Megaptera novaeangliae*

NOAA (2015b) reported that the Humpback whales can accumulate halogenated organic pollutants (including dichloro-diphenyl-trichloroethane - DDT), polychlorinated biphenyls-PCBs, hexachlorocyclohexane (HCH) and chlordane (CH) insecticides and pesticides in their blubber, as a result of feeding on contaminated prey (bioaccumulation) or inhalation in areas of high contaminant concentrations (*e.g.*, regions of atmospheric deposition) (Barrie et al. 1992; Wania and Mackay 1993). Concentrations of organochlorine pesticides, heavy metals, and PCB's have been reported in humpback whale tissues from Canadian, United States, and Caribbean waters (Taruski et el. 1975). The amount of information on levels of contamination of humpback whales on Southern Hemisphere feeding grounds is very limited. Elfes et al. (2010) described the range and degree of organic contaminants accumulated in the blubber of humpback whales sampled on Northern Hemisphere feeding grounds. Concentrations were reported to be high in Southern California and Northern Gulf of Maine, with higher levels of PCBs, PBDEs, and CH insecticides reported in the North Atlantic Ocean (Gulf of Maine and Bay of Fundy) than in the North Pacific (California, Southeast Alaska, Aleutian Islands). The highest levels of DDT were found in whales feeding off Southern California. Elfes et al. study also reported a linear increase in PCB, DDT, and chlordane concentration with age of the whales sampled. Generally,

concentrations of these contaminants in baleen whale such as humpback whales were low relative to levels found in odontocetes (O'Shea and Brownell, 1994).

The concentrations of total DDT and total PCB in humpback females and calves were similar. The data on the proportions (%) of PCB congeners in blubber biopsy samples from humpback females and calves indicate that the patterns of PCB contamination are the same in the adult females and the young animals. Similar patterns were observed in humpback females and calves for the relative proportions of DDT and metabolites, as well as the proportions of HCH isomers.

NOAA (2015b) further indicated that the health effects of different doses of contaminants are currently unknown for humpback whales (Krahn et al. 2004). Contaminant levels have been proposed as a causative factor in lower reproductive rates found among humpback whales off Southern California (Steiger and Calambokidis 2000), but at present the threshold level for negative effects, and transfer rates to calves, are unknown for humpback whales. Although there has been substantial research on the identification and quantification of such contaminants on individual whales, no detectable effect from contaminants has been identified in baleen whales. There may be chronic, sub-lethal impacts that are currently unknown. The difficulty in identifying contaminants as a causative agent in humpback whale mortality and/or decreased fecundity led to conclusion that the severity of this threat was low in all regions, except where lack of data indicated a finding of unknown.

Concentrations and patterns of persistent contaminants (PCBs and organochlorine contaminants) in whales have been studied by Metcalfe et al. (2004). In this study, using biopsies results of blubber from female blue and humpback whales from the Gulf of St. Lawrence, the concentrations of organic contaminants were examined to see whether or not the differences between this two species are related to their diets. Blue whales feed exclusively on krill, while humpback whales feed heavily on small fish. Overall, there were no significant differences found in concentrations of organic contaminants found in blubber of two species, however the proportions of some PCB congeners, HCH isomers, and DDT and its metabolites were different in the two baleen whale species, reflecting differences in the diet of the two species.

A comparison of patterns of DDT and metabolites in female blue whales and humpback whales shows that DDT is present in higher proportions in the female blue whales. This trend is also seen with PCB congener patterns.

Diet is the most obvious difference between humpback and blue whales that could explain inter-species differences in the contaminant data. Blue whales feed exclusively on krill, while humpback whales feed heavily on small fish. The different patterns of PCB congeners and DDT metabolites in blue and humpback whales could reflect differences in the patterns of these contaminants in fish and krill.

In Summary, numerous scientific studies have been published which focus on the concentrations of organic contaminants in the marine waters as well as bioaccumulated concentrations by whales (primarily in their blubber). Researchers investigated differences in concentrations of organic contaminants bioaccumulated by different whale species with different feeding patterns (for example baleen vs toothed whales, baleen plankton-eaters vs baleen small fish-eaters). In addition, researchers well studied differences in bioaccumulation patterns of the organic

contaminants between males and females of the same species and compared bioaccumulation of contaminants in females and their young ones. Differences appear to be well understood and justification for differences is often provided.

However, there is a limited amount of research done to measure the actual concentrations of the specific contaminants in the whale's tissue. The number of contaminants investigated is also very limited. Concentrations of contaminants which are being investigated (primarily PCBs and DDT) greatly vary not only among different whale species but also among different samples within the same species, often explained by different ambient water conditions. As it was pointed out by numerous researchers, at this point, there is not enough data to make any firm conclusions as to specific effects of the individual organic contaminants bioaccumulated by whales on their overall condition. The "safe" levels of specific organic contaminants are not yet established for any of whale species.

There is a general theory among many researchers that the organic contaminants which are being bioaccumulated in whale's tissues (especially organochlorines), severely inhibit the reproductive processes of whales and can lead to infertility, birth defects, tumors and lesions, tooth decay, adrenal gland cysts, a high %age of malignant growths. Although some of the contaminants were identified as a primary suspects, and their accumulated concentrations were quantified via biopsies, there was no relationship provided to further evaluate the impact of their concentrations in ambient waters. On the other hand, however, numerous researchers are pointing out that although the organic contaminants are found in whale's tissue, there is no apparent evidence that they are harmful to whales at all, and if they are, to what degree.

In summary, there is no information in the scientific literature indicating that the organic contaminants being accumulated by whales pose a threat to the survival and recovery of the whale populations. Concentrations of the organic contaminants, which would be considered "safe" to whales are not yet established. Their impacts on whale conditions are simply not yet known.

Conclusion: The EPA did not find any information which would suggest that concentrations of organic contaminants, adopted by the USVI as water quality criteria would pose a threat to the recovery of any of the ESA-listed whales. As a result, the EPA has determined that the water quality criteria adopted for the parameters listed below are NLAA the recovery of any of the ESA-listed whales or their habitats: pesticides (Aldrin, Carbaryl, Chlordane, Chlorpyrifos, Demeton, Diazinon, Dieldrin, Endosulfan s, Endrin, Lindane, Guthion, Heptachlor, Heptachlor Epoxide, Malathion, Methoxychlor, Mirex, Pentachlorophenol, Toxaphene, and 4,4'DDT) , Nonylphenol, Tributyltin and Polychlorinated biphenyls.

As soon as the additional information related to the toxicity of the above listed organic compounds on ESA-listed whale species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of all of the ESA-listed species.

- Hawksbill sea turtle, *Eretmochelys imbricata* and *Green sea turtle, Chelonia mydas*

As indicated by NOAA and FWS (1998), Hawksbills and Green turtles depend upon sea grass and/or coral reef habitats for food and habitat. The destruction or degradation of these sensitive ecosystems is a direct threat to the recovery of both ESA-listed species. The general degradation of these habitats can be affected by "chemical poisoning". Organic chemical pollutants, such as petroleum, sewage, pesticides, solvents, industrial discharges, and agricultural runoff are responsible for sea turtle mortality each year. Environmental contamination also harms biologically important nearshore ecosystems, including seagrass, coral, mangrove, and algae communities. The declining productivity of seagrass and coral communities, in particular, can be hazardous to sea turtles that depend on these systems for nutrition and shelter. However, as NOAA indicated, little is known about threats to foraging populations of hawksbills.

Impact on the organic compounds on the sensitive coral reef ecosystems was already evaluated in detail in the earlier sections of this document. The concentrations of all of the organic compounds of interest, which were adopted by the USVI as water quality criteria were determined NLAA the recovery of sea turtle species.

- *Leatherback sea turtle, Dermochelys coriacea*

As indicated by NOAA and FWS (1998), the chemical contamination of the Leatherbacks marine environment due to sewage, agricultural runoff, pesticides, solvents and industrial discharges is widespread along the coastal waters of the western United States. However, the impact of these contaminants on leatherbacks is still unknown.

- *Loggerhead sea turtle, Caretta caretta;*

As indicated by NOAA and FWS (2008) pollution sources that may most likely affect sea turtles include persistent chlorinated hydrocarbons and heavy metals. Widespread and persistent organochlorine contaminants, such as PCBs and pesticides, are known to have broad-ranging toxicities in wildlife. Long-lived carnivorous species, such as loggerheads, would tend to bioaccumulate these compounds. Multiple researchers indicate that the concentrations of organochlorine found in sea turtles have been, in general, much lower than those found in marine mammals probably due to the much lower metabolic rates of sea turtles. The impacts of these compounds have been shown to have deleterious effects on loggerheads (Keller et al. 2004).

Keller et al. (2004) investigated possible health effects of organochlorine contaminants on loggerhead sea turtles. Nonlethal fat biopsies and blood samples were collected from live turtles for organochlorine contaminant analysis, and concentrations were compared with clinical health assessment data, including hematology, plasma chemistry, and body condition. Researchers found that widespread and persistent organochlorine contaminants, such as polychlorinated biphenyls (PCBs) and pesticides, may be affecting the health of loggerheads even though sea turtles accumulate lower concentrations of organochlorine contaminants compared with other wildlife. Concentrations of total PCBs, DDTs, chlordanes, dieldrin, and mirex were determined

in 44 fat biopsies and 48 blood samples. Blood concentrations of chlordanes were negatively correlated with hemoglobin, indicative of anemia. Positive correlations were observed between most contaminants and white blood cell counts and between mirex and PCB concentrations, suggesting modulation of the immune system. All classes of contaminants in the blood (except dieldrin) were correlated positively with aspartate aminotransferase activity, indicating possible hepatocellular damage. Significant correlations to levels of certain contaminant classes also suggested possible alteration of protein, carbohydrate and ion regulation. Keller et al. (2004) found significant correlations for a wide variety of biological functions, suggesting, for example, changes in the immune system, possible liver damage, and possible alterations in protein and carbohydrate regulation. However, the authors cautioned that the correlations suggest, but do not prove, a cause and effect link. The specific links to the levels of these compounds in the ambient waters are yet to be determined.

Metals and persistent organic contaminants are globally present in aquatic systems and their potential transfer to loggerhead turtles has become a serious threat for their health status. D'llio at al. (2011) published an overview of the international studies carried out on the quantification of metals, polychlorinated biphenyls (PCBs) and organochlorines in tissues, organs and fluids of loggerhead turtles from the Mediterranean Sea, the Atlantic and the Pacific Oceans.The environmental fate of these contaminants was traced by researchers by the analysis of turtles' tissues and blood. Generally, loggerhead turtles exhibited a higher metal load than other turtle species, this could be explained by differences in diet habits being food the main source of exposure. Literature shows that muscle, liver and kidney are most considered for the quantification of chemical elements, while, organic compounds are typically investigated in liver and fat.

Mckenzie et al. (1999) evaluated the concentrations of individual polychlorobiphenyls (PCBs) and organochlorine pesticides in marine turtle tissues collected from the Mediterranean and European Atlantic waters between 1994 and 1996. The PCB concentrations were highest in adipose tissue and ranged from 775 to 893 in loggerhead, 39 to 261 in green and 47 to 178 μg/kg wet wt in leatherback turtles. Omnivorous loggerhead turtles had the highest organochlorine contaminant concentrations in all tissues sampled. It is thought that dietary preferences were likely to be the main differentiating factor among species. Decreasing lipid contaminant burdens with turtle size were observed in green turtles, most likely attributable to a change in diet with age.

In Summary, numerous scientific studies have been published which focus on the concentrations of organic contaminants in the marine waters as well as concentrations bioaccumulated by sea turtles. Researchers investigated differences in the distribution and concentrations of organic contaminants bioaccumulated by different sea turtle species.

Many researchers focused their studies to measure the actual concentrations of the specific contaminants in the turtle tissue. At this point, we have a significant amount of information on how much of the individual contaminants are being bioaccumulated and at which locations in the body. The amount of studies trying to link these compounds to the health effects of the sea turtles are very limited. These studies, although very informative do not provide a link between potential health problems with the amount s of the individual contaminants in the turtles bodies

and furthermore, do not provide a relationship to the amount of these contaminants present in the ambient waters.

As it was pointed out by numerous researchers, at this point, there is not enough data to make any firm conclusions as to specific effects of the individual organic contaminants bioaccumulated by sea turtles on their overall condition. The "safe" levels of specific organic contaminants are not yet established for any of sea turtle species.

In summary, there is no information in the scientific literature indicating that the organic contaminants being accumulated by sea turtles actually pose a threat to the survival and recovery of their populations. Concentrations of the organic contaminants, which would be considered "safe" to sea turtles are not yet established. We have just begun to learn about their impacts on sea turtles conditions.

Conclusion: The EPA did not find any information which would suggest that concentrations of organic contaminants, adopted by the USVI as water quality criteria would pose a threat to the recovery of any of the ESA-listed sea turtle species. As a result, the EPA has determined that the water quality criteria adopted for the parameters listed below are NLAA the recovery of any of the ESA-listed turtles or their habitats: pesticides (Aldrin, Carbaryl, Chlordane, Chlorpyrifos, Demeton, Diazinon, Dieldrin, Endosulfan s, Endrin, Lindane, Guthion, Heptachlor, Heptachlor Epoxide, Malathion, Methoxychlor, Mirex, Pentachlorophenol, Toxaphene, and 4,4'DDT) , Nonylphenol, Tributyltin and Polychlorinated biphenyls.

As soon as the additional information related to the toxicity of the above listed organic compounds on ESA-listed turtle species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of all of the ESA-listed species.

c) Analysis of potential impacts of organic compounds on Fish

- Scalloped Hammerhead Shark, *Sphyrna lewini*

Lyons and Dauglass (2015) demonstrated that scalloped hammerheads are exposed to a substantial amount of contaminants even before they are born. This is a result of the maternal offloading processes in sharks, which represents a substantial source of exposure for young sharks and is a significant pathway for contaminant redistribution within marine ecosystems (Lyons and Douglass 2015). Comparable to mammalian systems, scalloped hammerhead sharks utilize a yolk-sac placental strategy to nourish young in utero, which may allow females to transfer contaminants to young. Lyons and Dauglass measured organic contaminants such as PCBs and chlorinated pesticides in livers of two near-term pregnant females and their embryos. Adult female hammerheads and embryos from the two litters had similar levels of organic contaminant concentrations. The potential health impacts of these transferred contaminants are however unknown.

Although multiple researchers evaluated concentrations of the organic contaminants bioaccumulated in sharks as well as their distribution in the body, there were no studies identified in the scientific review process which would provide a link to the health effects of these contaminants on sharks. The EPA came across multiple studies which focused on "contaminations: present in sharks mussels, as they relate to sharks as a human prey. This perspective, however is outside of the scope for this BE.

- Nassau Grouper, *Epinephelus striatus*

Similar to Hawksbills and Green turtles, Nassau grouper also depend upon sea grass and/or coral reef habitats for food and habitat. The destruction or degradation of these sensitive ecosystems is a direct threat to the recovery of this species. The general degradation of these habitats can be affected by "chemical poisoning" through exposure to petroleum, sewage, pesticides, solvents, industrial discharges, and agricultural runoff caring pesticides. Environmental contamination also harms biologically important nearshore ecosystems, including seagrass, coral, mangrove, and algae communities. The declining productivity of seagrass and coral communities, in particular, can be hazardous to Nassau grouper that depend on these systems for nutrition and shelter.

Impact on the organic compounds on the sensitive coral reef ecosystems was already evaluated in detail in the earlier sections of this document. The concentrations of all of the organic compounds of interest, which were adopted by the USVI as water quality criteria were determined NLAA the recovery of corals, providing safe fish habitat.

In summary, numerous scientific studies have been published which focus on the concentrations of organic contaminants in the marine waters as well as concentrations bioaccumulated by fish. Researchers investigated differences in the distribution and concentrations of organic contaminants bioaccumulated by both ESA-listed fish species.

Many researchers focused their studies to measure the actual concentrations of the specific contaminants in the fish tissue, as a concern for potential effect on humans. The amount of studies trying to link these compounds to the health effects of fish is nonexistent. As a result, there is not enough data to make any firm conclusions as to specific effects of the individual organic contaminants bioaccumulated by fish on their overall condition. The "safe" levels of specific organic contaminants are not yet established for any of fish species.

In summary, there is no information in the scientific literature indicating that the organic contaminants being accumulated by fish actually pose a threat to the survival and recovery of their populations. Concentrations of the organic contaminants, which would be considered "safe" to fish are not yet established.

Conclusion: The EPA did not find any information which would suggest that concentrations of organic contaminants, adopted by the USVI as water quality criteria would pose a threat to the recovery of any of the ESA-listed fish species. As a result, the EPA has determined that the water quality criteria adopted for the parameters listed below are NLAA the recovery of any of the ESA-listed fish species or their habitats: pesticides (Aldrin, Carbaryl, Chlordane, Chlorpyrifos, Demeton, Diazinon, Dieldrin, Endosulfan s, Endrin, Lindane, Guthion, Heptachlor,

Heptachlor Epoxide, Malathion, Methoxychlor, Mirex, Pentachlorophenol, Toxaphene, and 4,4'DDT) , Nonylphenol, Tributyltin and Polychlorinated biphenyls.

As soon as the additional information related to the toxicity of the above listed organic compounds on ESA-listed fish species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of all of the ESA-listed species.

8. Toxic Pollutants - Inorganic Compounds

a) Analysis of potential impacts of inorganic compounds on Corals

(1) Metals

(a) Arsenic

During the s cientific literature review, the EPA did not come across any research which would report on the specific effects of arsenic on ESA-listed corals. In addition, the EPA did not find any information which would suggest that arsenic concentrations of 69 µg/L or below would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for arsenic of 69 µg/L (as acute value) and 36 µg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

As soon as the additional information related to the toxicity of arsenic on coral species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of the sensitive coral reef ecosystems.

(b) Cadmium

Reichelt-Brushett and Harrison (1999) evaluated the toxic effects of cadmium on the fertilization rates of gametes from the scleractinian coral *Goniastrea aspera*. Spawned eggs and sperm were collected from adult colonies of *G. aspera* and dosed separately with different concentrations of copper, zinc or cadmium, with normal seawater used as controls. The eggs and sperm were then combined to allow fertilization to occur. After 5 hours development time, the number of fertilized and developing embryos and unfertilized eggs were counted and recorded to determine %age fertilization. Cadmium did not affect fertilization success at the concentrations tested. High fertilization rates were recorded in controls and in all concentrations of cadmium up to 200 µg/L. Additional work on gametes from the reef coral *Oxypora lacera* showed no decrease in fertilization success up to 1000 µg/L of cadmium.

Reichelt-Brushett and Harrison (2005) provided information on the effects of various concentrations of the trace metals copper, lead, zinc, cadmium, and nickel on fertilization success of gametes from the scleractinian reef corals *Goniastrea aspera*, *Goniastrea retiformis*, *Acropora tenuis*, and *Acropora longicyathus*. The EC50 values (the concentration that reduces the fertilization rate by 50% relative to the control fertilization) for copper effects on fertilization success of these coral species range from 15 to 40µg/L. The EC50 values for lead were 1450–1800 µg/L for the *Acropora* species, and >2400µg/L for *G. aspera* gametes. Fertilization responses to zinc and nickel were variable and a significant reduction in fertilization success for *A. tenuis* gametes was found only at very high cadmium concentrations.

Mitchelmore et al. (2007) exposed fragments of *Pocillopora damicornis* in the laboratory to three different concentrations of cadmium chlorine (0, 5 and 50µg/L) for 14 days and analyzed for metal content in the whole association, algal or animal fractions. Various physiological and biochemical parameters were also measured, such as, algal cell counts, mitotic index, chlorophyll content and levels of the antioxidant glutathione (GSH). Cadmium accumulations were observed at all time points and doses; there was no evidence of differential metal partitioning between the algal or animal fractions. No changes in algal cell density, mitotic index or chlorophyll content from the controls were observed in any of the metal treatments. GSH levels were significantly higher in the 5µg/L cadmium treatment (day 4) compared with controls at the same time point. Although no evidence of a bleaching response occurred, corals in 50µg/L metal exposure sloughed off tissues and did not survive the duration of the exposure period. This research demonstrated the accumulation of cadmium in *P. damicornis* and mortality in the absence of a bleaching response.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of cadmium on ESA-listed corals. In addition, the EPA did not find any information which would suggest that cadmium concentrations of 40 µg/L or below would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for cadmium of 40 µg/L (as acute value) and 8.8 µg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of cadmium on coral species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(c) Chromium (VI)

During the scientific literature review, the EPA did not come across any research which reported on the specific effects of chromium (VI) on ESA-listed corals. In addition, the EPA did not find any information which would suggest that chromium (VI) concentrations of 1,100 µg/L or below would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for chromium VI of 1,100 µg/L (as acute value) and 50 µg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

As soon as the additional information related to the toxicity of chromium VI on coral species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of the sensitive coral reef ecosystems.

(d) Copper

Harrison (1994) quantified the effects of various concentrations of copper on fertilization success of gametes from the brain coral *Goniastrea aspera*. Copper was toxic to the fertilization of *G. aspera* gametes at concentrations of 20 μg/L and above. Copper concentrations of 2 μg/L resulted in mean fertilization rates of 93% ± 4.03% with *G. aspera* gametes, which is not significantly different to the controls at 91% ± 3.23%. However, the higher copper concentrations tested, resulted in significant reductions in fertilization rates. Mean fertilization success was significantly reduced to 41% ± 7.12% (45% of the controls) with a dose of 20 μg/L copper for 5 h, and was <1% at 200 μg/L copper. The EC50 value for fertilization of *G. aspera* gametes exposed to copper was 14.5 μg/L.

Reichelt-Brushett and Harrison (1999) evaluated the toxic effects of copper on the fertilization success of scleractinian coral *Goniastrea aspera* gametes. Spawned eggs and sperm were collected from adult colonies of *G. aspera* and dosed separately with different concentrations of copper with normal *seawater* used as controls. The eggs and sperm were then combined to allow fertilization to occur. After 5 h development time, the number of fertilized and developing embryos and unfertilized eggs were counted and recorded to determine %age fertilization. High fertilization rates of 91% ± 3.2% and 93% ± 4.0% were recorded in the copper controls and in 2 μg/L of copper, respectively. However, fertilization success was significantly reduced to 41% ± 7.1% at 20 μg/L of copper, and <1% fertilization occurred at 200 μg/L of copper. These data are the first to show that relatively low concentrations of copper significantly affect fertilization rates of spawned gametes of reef corals from the Great Barrier Reef.

Reichelt-Brushett and Harrison (2000) examined the effect of copper on the settlement success of planula larvae of the reef-building coral *Acropora tenuis* during 1994 and 1996 at Magnetic Island, Great Barrier Reef. Copper concentrations of 2, 10, 20 μg/L did not inhibit larval settlement after 48-hours exposure. However, copper concentrations of 42 μg/L and 81 μg /L significantly reduced settlement success of *A. tenuis* larvae after 48-hours exposure compared with controls using normal seawater. At copper concentration of 200 μg/L all larvae were reported dead. EC50 values for the effect of copper on *A. tenuis* larval settlement were calculated from the 1996 results using measured copper concentrations. The 48-h EC50 was 35 μg /L with an upper and lower 95% confidence limit of 37 μg /L and 32 μg /L, respectively. The 48-hours NOEC value for both experiments was 20 μg /L copper. These experiments provide some of the first data on sub-lethal effects of trace metals on tropical marine organisms, and demonstrate that relatively low copper concentrations impair or inhibit settlement of coral larvae.

Reichelt-Brushett and Harrison (2005) provided additional information on the effects of various concentrations of the trace metals including copper on fertilization success of gametes from the scleractinian reef corals *Goniastrea aspera*, *Goniastrea retiformis*, *Acropora tenuis*, and *Acropora longicyathus*. The EC50 values (the concentration that reduces the fertilization rate by

50% relative to the control fertilization) for copper effects on fertilization success of these coral species range from 15 to 40 μg/L.

Mitchelmore et al. (2007) exposed fragments of *Pocillopora damicornis* in the laboratory to copper chlorides (0, 5 and 50μg/L) for 14 days and analyzed for metal content in the whole association, algal or animal fractions. Various physiological and biochemical parameters were also measured, such as, algal cell counts, mitotic index, chlorophyll content and levels of the antioxidant glutathione (GSH). Copper accumulations were observed at all-time points and doses; there was no evidence of differential metal partitioning between the algal or animal fractions. No changes in algal cell density, mitotic index or chlorophyll content from the controls were observed in any of the metal treatments. GSH levels were significantly higher in the 5μg/L copper treatments (days 4 and 14) compared with controls at the same time point. Although no evidence of a bleaching response occurred, corals in 50μg/L metal exposures sloughed off tissues and did not survive the duration of the exposure period. This research demonstrated the accumulation of copper in *P. damicornis* and mortality in the absence of a bleaching response.

Negri and Heyward (2000) evaluated inhibition of fertilization and larval metamorphosis of the coral *Acropora millepora* in response to solution of copper using laboratory-based bioassays. Nominal concentrations that inhibited 50% fertilization and metamorphosis (IC50) were calculated from 4-hours fertilization and 24-hours metamorphosis assays and were based on introduced dose. Copper inhibited fertilization in 50% of *A. millepora* gametes relative to control assays at a nominal concentration of 17.4 μg/L (IC50). Copper inhibited 50% metamorphosis at a concentration of 110 μg/L. These results for copper are consistent with other studies of its effects on coral fertilization. An IC50 of less than 100 μg/L was reported for both *Favites chinensis* and *Platygyra ryukyuensis* (Heyward 1988) and 14.5 μg/L for *Goniastrea aspera* (Reichelt-Brushett and Harrison 1999). A larval bioassay (96 h) for the coral *Pocillopora damicornis* yielded an LC50 of 63 μg/L (Esquivel 1986). Howard, Crosby and Porfirio (1986) reported the LC50 of copper to adult colonies of *Montipora verrucosa* as 48 μg/L. Copper is known to damage sperm of other species in various ways, such as oxidative stress (e.g. Loyd et al. 1997) but the mechanisms underlying coper toxicity to coral fertilization have not be demonstrated.

Alutoin et al (2001) studied the physiological effects on the hermatypic coral *Porites lutea* when exposed to a combination of reduced salinity (30 psu-20 psu ambient) and two concentrations of copper (10 μg/L and 30 μg/L). The physiological response was estimated by measuring net primary production rate and respiration per surface area. There were no significant effects on respiration rate in any of the treatments when compared to controls or between each other. Exposure to 30 μg/L of copper and reduced salinity significantly reduced production rate and chlorophyll *a* concentrations while corals exposed to 10 μg/L of copper remained unaffected.

Esquivel (1986) performed 96-hours static acute copper toxicity studies on the planula of the reef coral *Pocillopora damicornis*, with LC50 of 0.067 mg/L. At copper concentrations ranging from 0.01 to 0.1 mg/L, symbiotic algae were expelled, and mucus was produced.

Sabdono (2009) quantified heavy metal concentration in the tissue of coral *Galaxea fascicularis*and evaluated the toxic effect of metal on coral. Series of copper exposures at

concentrations of 0.025, 0.05, 0.075 and 0.1 mg/L were conducted for 96-hours. Short duration (24hours) laboratory assay demonstrated coral tissue bleaching and death at copper concentrations of 0.1 mg/L. The 96-hours LC50 was determined to be 0.032 mg/L. The results of this study were similar to the earlier findings of Howard et al (1986) who reported a 96-hours LC50 for *Montipora verucose* exposed to 0.048 mg/L of copper. While Jones (1997) demonstrated the loss of zooxanthellae from *Acropora formosa* exposed at 0.01 to 0.04 mg/L of copper for 48 hours. Mitchelmore et al. (2007) showed 0.05 mg/L exposure to copper to cause severe stress for *P. damicornis*.

Negri and Heyward (2001) assessed inhibition of fertilization and larval metamorphosis of the coral *Acropora millepora* in response to copper solutions using laboratory-based bioassays. Nominal concentrations that inhibited 50% fertilization and metamorphosis (IC50) were calculated from 4 hour fertilization and 24 hour metamorphosis assays and were based on introduced dose. Copper was potent towards fertilization with an IC50 of 17.4 μg/L. Copper inhibited fertilization in 50% of *Acropora millepora* gametes relative to control assays at a nominal concentration of 17.4 μg/L. Copper inhibited 50% metamorphosis at a concentration of 110.2 μg/L.

These results for are consistent with earlier findings for coral fertilization as it relates to copper. An IC50 of less than 100 μg/L was reported for both *Favites chinensis* and *Platygyra ryukyuensis* (Heyward 1988) and 14.5 μg/L for *Goniastrea aspera* (Reichelt-Brushett & Harrison 1999). A larval bioassay (96 h) for the coral *Pocillopora damicornis* yielded an LC50 of 63 μg/L (Esquivel 1986). Howard et al. (1986) reported the LC50 of copper to adult colonies of *Montipora verrucosa* as 48 μg/L.

Conclusion: Although copper appears to be the most researched metal, as related to the effects on corals, during the scientific literature review, the EPA did not come across sufficient research which would report on the specific effects of copper on all of the ESA-listed corals. In addition, the EPA did not find any information which would suggest that copper concentrations of 4.8 μg/L or below would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for copper of 4.8 μg/L (as acute value) and 3.1 μg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of copper on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(e) Lead

Reichelt-Brushett and Harrison (2005) provided information on the effects of various concentrations of the trace metals including lead on fertilization success of gametes from the scleractinian reef corals *Goniastrea aspera*, *Goniastrea retiformis*, *Acropora tenuis*, and *Acropora longicyathus*. The EC50 values (the concentration that reduces the fertilization rate by 50% relative to the control fertilization) for lead effects on fertilization success of these coral

species range from 1450 to 1800 µg/Lfor the *Acropora* species, and >2400 µg/L for *G. aspera* gametes. These concentrations are significantly higher when compared to values adopted by the USVI as criteria.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of lead on all of the ESA-listed corals. In addition, the EPA did not find any information which would suggest that lead concentrations of 210 µg/L or below would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for lead of 210 µg/L (as acute value) and 8.1 µg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of lead on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(f) Mercury – Methylmercury

Bastidas and Garcia (2004) conducted semi-static, chronic bioassays using HgCl2 on specimens of *Porites asteroides* collected from Bajo Caiman, Venezuela. These experiments examined the concentration of mercury in adult coral colonies; the distribution of Hg throughout three coral compartments (polyps, zooxanthellae and skeleton); and the sublethal effects of mercury on *P. asteroides*. To study the effects of sublethal doses of mercury on corals, colonies of *P. astreoides* were exposed to nominal mercury concentrations of 0.01, 0.1 and 0.5 mg/L using semi-static, chronic bioassays for up to 15 days, with HgCl2 administered by pulses every 3 days (mean Hg concentration in the water was 0.004, 0.037 and 0.180 mg/L, respectively). Mercury concentration in the water was variable due to natural losses and the pulsed doses given during the assays. While total mercury in the corals was directly proportional to mercury exposure, analysis of the different coral compartments (polyps, zooxanthellae and skeleton) showed that zooxanthellae and the skeleton accumulated mercury in direct relation to Hg exposure, but polyp tissue accumulated more mercury at 0.1 than at 0.5 mg/L. These results supported the hypothesis that coral polyps may have actively diverted mercury to other coral compartments as a method of detoxification. Absolute mercury concentration values (per unit surface area) were highest in the zooxanthellae, followed by polyps, and then the skeleton. Colonies exposed to the highest mercury concentration accumulated 89% of which was found in zooxanthellae, 7% in polyps and 4% in the skeleton. The capacity of zooxanthellae and the skeleton to concentrate mercury and the decrease in zooxanthellae density support the hypothesis that polyps may divert mercury to these 2 coral compartments as a detoxifying mechanism.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of mercury on any of the ESA-listed corals. In addition, the EPA did not find any information which would suggest that mercury concentrations of 1.8 µg/L or below would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for

mercury of 1.8 μg/L (as acute value) and 0.94 μg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of mercury on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(g) Nickel

The only study found to be relative to effects of nickel on corals was one published by Goh (1991), who studied the effects of nickel on the mortality and settlement of planula larvae from the brooding coral *Pocillopora damicornis*, and found that nickel concentrations of 1000 μg/L significantly affected survival and settlement of the larvae. Results indicated that a nickel concentration of 9 mg/L over 12 hours exposure was sufficient to cause 50% mortality in larvae 39.6 hours after the removal of the toxicant. The 12 hours LC50 for nickel was reported to be 9 mg/L. Settlement in larvae was more sensitive, showing significantly reduced settlement rates from 9 days into recovery, after exposure to 1 mg/L of nickel at durations of 12 to 96 hours.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of nickel on any of the ESA-listed corals. In addition, the EPA did not find any information which would suggest that nickel concentrations of 74 μg/L or below would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for nickel of 74 μg/L (as acute value) and 8.2 μg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of nickel on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(h) Selenium

During the scientific literature review, the EPA did not come across any research which would report on the specific effects of selenium on ESA-listed corals. In addition, the EPA did not find any information which would suggest that selenium concentrations of 290 μg/L or below would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for selenium of 290 μg/L (as acute value) and 71 μg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

As soon as the additional information related to the toxicity of selenium on coral species becomes available, the Agency will reevaluate its determination and work with the USVI to

revise the applicable water quality criteria, if needed, to ensure that they are fully protective of the sensitive coral reef ecosystems.

(i) Silver

During the scientific literature review, the EPA did not come across any research which would report on the specific effects of silver on ESA-listed corals. In addition, the EPA did not find any information which would suggest that silver concentration of 1.9 μg/L would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criterion adopted by the USVI for silver of 1.9 μg/L (as acute value) is NLAA the recovery of seven ESA-listed coral species or their habitats.

As soon as the additional information related to the toxicity of silver on coral species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of the sensitive coral reef ecosystems.

(j) Zinc

Reichelt-Brushett and Harrison (1999) evaluated fertilization success of gametes from the scleractinian coral *Goniastrea aspera* to determine toxic effects of the trace metals including zinc. Spawned eggs and sperm were collected from adult colonies of *G. aspera* and dosed separately with different concentrations of zinc, with normal seawater used as controls. The eggs and sperm were then combined to allow fertilization to occur. After 5 hours development time, the number of fertilized and developing embryos and unfertilized eggs were counted and recorded to determine %age fertilization. Zinc did not affect fertilization success at any of the concentrations tested. High fertilization rates were recorded in controls and in all concentrations of zinc up to 500 μg/L.

Heyward (1988) studied the effect of zinc sulphates on fertilization rates in *Favites chinensis* and *Platygyra ryukyuensis*. He found that zinc concentrations of 1000 μg/L inhibited fertilization success for gametes of both species.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of zinc on any of the ESA-listed corals. In addition, the EPA did not find any information which would suggest that zinc concentrations of 90 μg/L or below would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for zinc of 90 μg/L (as acute value) and 81 μg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of zinc on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(2) Non-metals:

(a) Chlorine

Best et al. (1981) tested the effects of 0.49 mg/L chlorine on the planulae of three species of Hawaiian corals, and found that exposure for up to 7 hours was not lethal. In 96 hours static bioassays, the growth rates of the phytoplankters *Chaetoceros gracilis* and *Dunaliella tertiolecta* were various levels of sedimentation reduced by 50 % when exposed to initial concentrations of 0.09 to 0.32 mg/L of oxidants induced by addition of chlorine. The LC50 for larvae of the coral reef urchin *Echinometra rnathaei* ranged from 0.46 to 0.84 mg/L of chlorine-induced oxidants, whereas the LC50 for veligers of *Stylocheilus longicauda* was over 1.95 mg/L.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of chlorine on any of the ESA-listed corals. In addition, the EPA did not find any information which would suggest that chlorine concentrations of 13 μg/L or below would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for chlorine of 13 μg/L (as acute value) and 7.5 μg/L (as chronic value) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of chlorine on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(b) Ammonia

During the scientific literature review, the EPA did not come across any research which would report on the specific effects of ammonia on ESA-listed corals. In addition, the EPA did not find any information which would suggest that ammonia concentration adopted by the USVI as water quality criteria would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for ammonia is NLAA the recovery of seven ESA-listed coral species or their habitats.

As soon as the additional information related to the toxicity of ammonia on coral species becomes available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of the sensitive coral reef ecosystems.

(c) Cyanide

Cyanide fishing, which involves spraying or dumping cyanide onto reefs to stun and capture (live) fish, kills coral polyps and degrades the reef habitat. Exposure of corals to cyanide can result in a reduction or cessation of respiration, a reduction in phototrophic potential and a

decrease in growth rates and fecundity. The most obvious response is bleaching. Re-establishment of the symbiosis may take from six months to one year or more.

Ross and Hoegh-Guldberg (1999) exposed *Plesiastrea versipora* to various concentrations of cyanide under different light intensities in several experiments conducted over 3 months. Collectively, these results suggest that cyanide causes the dissociation of the coral-algal symbiosis by affecting photosynthesis of the zooxanthellae as opposed to host or symbiont respiration.

Because cyanide dissociates rapidly in the seawater, its effects are limited to the immediate vicinity (Clark 2001). Cyanide was reported to induce coral bleaching (Jones and Hoegh-Guldberg 1999 and Cervino et al. 2003).A three-hour exposure to cyanide at concentration as low as 10 uM was found to impair photosynthesis and induced sub-lethal bleaching (Jones and Hoegh-Guldberg 1999). Corals exposed to 100 uM cyanide lost 60% of their algal symbionts and bleached overtime. Short (1 to 2 minutes) exposures to concentrations between 2 to 22 uM cyanide also caused variable bleaching and mortality in corals (Cervino et al. 2003), while higher exposure to cyanide of 200 uM for 10 minutes or 10 uM for 3hours, caused total mortality. As a result, cyanide- induced bleaching occurs at low concentrations (10 uM) and the severity of the response increases with higher concentrations and higher exposures.

In a laboratory-based study, Jones & Steven (1997) showed that brief exposure to elevated cyanide concentration caused the corals *Pocillopora damicornis* and *Porites lichen* to lose their symbiotic photosynthetic algae (zooxanthellae). Similar loss of zooxanthellae from corals has been observed in response to variation in a wide range of physical and chemical parameters (Brown and Howard 1985, Hoegh- Guldberg & Smith 1989, Jones 1997). Loss of zooxanthellae causes corals to bleach, which is usually associated with the discoloration of corals following periods of elevated seawater temperatures.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of cyanide on any of the ESA-listed corals. In addition, the EPA did not find any information which would suggest that cyanide concentration of 1µg/L would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for cyanide of 1µg/L (as both, acute and chronic values) are NLAA the recovery of seven ESA-listed coral species or their habitats.

Should additional information related to the effects of cyanide on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

(d) Sulfide-hydrogen sulfide

During the scientific literature review, the EPA did not come across any research which would report on the specific effects of sulfide on ESA-listed corals. In addition, the EPA did not find any information which would suggest that sulfide concentration adopted by the USVI as water

quality criteria would pose a threat to the recovery of any of the ESA-listed coral species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for sulfide of 2 μg/L (as a chronic value) is not likely to adversely affect (NLAA) the recovery of seven ESA-listed coral species or their habitats.

As soon as the additional information related to the toxicity of sulfide on coral species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of the sensitive coral reef ecosystems.

b) Analysis of potential impacts of inorganic compounds on Whales

(1) Metals

- Fin whale, *Balaenoptera physalus*

Sanpera et al. (1996) analyzed the concentrations of cadmium, copper and zinc in muscle, liver and kidney tissues of fin whales (*Balaenoptera physalus*) from two locations in the North Atlantic, Iceland and Spain. The concentrations of zinc in the muscle and that of cadmium in the liver and the kidney were significantly higher in fin whales from Iceland. Other differences between whales from the two areas concern the dynamics of cadmium in the organism. These findings support the hypothesis that fin whales from the two sites belong to different stocks and that cadmium in the organism can be used as a complementary tool in studies of population identity.

- Sei whale, *Balaenoptera borealis*

As stated by NOAA (2011a), based on studies of contaminants in baleen whales, including sei whales, pollutants do not appear to be a major threat to sei whales in most areas where sei whales are found. O'Shea and Brownell (1994) indicated that concentrations of metals in tissues of baleen whales were low, and lower than other marine mammal species. They further stated that there was no firm evidence that levels of heavy metals in baleen whales generally were high enough to cause toxic or other damaging effects.

The sei whale's strong preference for copepods and euphausiids (i.e., low trophic level organisms), at least in the North Atlantic, may make it less susceptible to the bioaccumulation of metal contaminants than, for example, fin, humpback, and minke whales, all of which seem to feed more regularly on fish and euphausiids (O'Shea and Brownell 1994). Because sei whales off California often feed on pelagic fish as well as invertebrates (Rice 1977), they might accumulate contaminants to a greater degree than do sei whales in the North Atlantic. There is no evidence that levels of heavy metals in baleen whales generally (including sei whales) are high enough to cause toxic or other damaging effects (O'Shea and Brownell 1994). It should be emphasized, however, that very little is known about the possible long-term and trans-

generational effects of exposure to pollutants, or about the possible compounding effects of exposure to two or more pollutants, in virtually any marine mammal species.

- Humpback whale, *Megaptera novaeangliae*
- Sperm whale, *Physeter macrocephalus*

Global baseline data on several metal contaminants were collected from apparently healthy sperm whales during cruises aboard the research vessel *Odyssey* conducted between 1999 and 2005 (Wise et al. 2009; Savery et al 2014). Some of the metals analyzed included silver, lead, chromium, mercury, selenium, and arsenic.

Silver is a pollutant that is increasingly used in nanotechnology and is of concern in marine waters due to its chemistry, toxicity, and bioavailability. Sperm whales are likely exposed to silver through consumption of prey and swallowing sea water. Silver is a potent inhibitor of selenium, which is an essential element that functions as an antioxidant in cells. Thus, in areas with elevated silver levels, sperm whales may become more sensitive to the toxicity of other chemicals or diseases. Silver levels in sperm whale skin samples were generally low. However, the apparent lack of accumulation may be a consequence of not being able to age the whales more accurately or small sample sizes per region. Researches notes that more research is needed to understand levels at which negative reproductive or physiological consequences are experienced.

Lead is prevalent in the environment and more than 90% of its presence is a result of anthropogenic activities. Lead has a high potential for bioaccumulation and serious health consequences. Sperm whales are likely exposed to lead through consumption of prey, swallowing water, or accidental consumption of sediment during bottom feeding (Savery et al. 2014). Lead concentrations in sperm whale skin were high in Papua New Guinea, the Bahamas, Australia, and several areas in the Sea of Cortez. Sources of the elevated lead concentrations may be from mining operations (Savery et al. 2014).

Chromium is released into the marine environment by both natural (primarily continental dust) and anthropogenic sources. Hexavalent chromium (Cr (VI)) is the predominant form of chromium found in marine waters and can have residence times up to 18 years (Wise et al. 2009). However, Cr (VI) degrades rapidly to trivalent chromium in marine organisms. Thus, studies typically measure total chromium levels in marine organisms. Overall, Wise et al. (2009) found high levels of chromium in sperm whales, but the consequence of such high levels is not known. The toxicity of CR (VI) in whales is also not well understood (Wise et al. 2011).

Mercury is of high concern due to its toxicity, stability in the atmosphere and ability to bioaccumulate throughout food webs (Savery et al. 2013). Mercury is released to the atmosphere from natural and anthropogenic sources (Pacyna et al. 2006). Of the populations studied, sperm whales from the Mediterranean Sea had the highest concentration of mercury. The Mediterranean Sea is thought to have high mercury levels due to geological, biochemical, and or ecological processes in the area (Savery et al. 2013).

Selenium, an essential element in marine mammal tissue, binds with mercury and may have a protective effect against mercury's toxicity. Selenium concentrations in sperm whales in this area were much higher than mercury suggesting that mercury is being detoxified in the skin tissue by some other element (Savery et al. 2013).

Arsenic is among the top 10 most hazardous substances in the environment based on toxicity and potential of exposure from air, water, and soil (Savery et al. 2013). It occurs naturally in the earth's crust and is also introduced through anthropogenic sources such as emissions from smelters or coal-fired power plants. By ocean basin, the highest mean arsenic concentration was found in sperm whales in the Indian Ocean particularly from the Seychelles, Maldives, and Sri Lanka. Overall, the global mean arsenic concentration for females was significantly lower than males possibly due to the female's ability to pass arsenic loads to their offspring during birth and lactation, as well as differences in hormone metabolism. Differences in mean arsenic concentration between sexes may also be a result from males eating larger prey and foraging deeper waters and coming in contact more frequently with the ocean benthos (Savery et al. 2013). The mean arsenic concentrations in sperm whales from this study were three-fold greater than concentrations found in other toothed whales. However, the arsenic found in sperm whales in this study was thought to be the nontoxic form, arsenobetaine.

Sperm whales in the Gulf of Mexico had significantly higher concentrations of nickel and chromium than the global mean average from the global surveys conducted in 1999 through 2005 (discussed above; Wise et al. 2009). The mean global nickel concentration was 2.4 ppm (n = 298; measured as μg/g wet weight and expressed as ppm). Whereas, in this study the average nickel concentration in the Gulf of Mexico sperm whales after the Deepwater Horizon was 15.9 ppm, which is 6.6 times higher than the global average (Wise et al. 2014). Also, resident females and immature males had higher nickel concentrations than the global mean, yet mature males that migrate beyond the Gulf of Mexico to forage in higher latitudes had similar values to the global mean.

Holsbeek et al. (1999) analyzed tissue samples obtained from 21 sperm whales that stranded in the North Sea in 1994/1995. Results indicate that mercury, PCB, DDE, and PAH levels were low and similar to levels reported for other marine mammals. However, cadmium levels were high, and double the reported levels in North Pacific sperm whales. While these strandings were not attributable to contaminant burdens, Holsbeek et al. (1999) suggested that the stable pollutants might affect the health or behavior of North Atlantic sperm whales.

As reported by NOAA (2010b), the Ocean Alliance, Inc. completed a five-year collection of baseline data on contaminants in the oceans. The team collected 955 sperm whale biopsy samples in 18 regions across the globe, with the goal of using sperm whales as global indicators of ocean contamination. Analysis of toxic metals contained in the samples revealed high levels of aluminum in all samples, with more significant levels in the Atlantic and Indian Oceans than in the Pacific Ocean or Mediterranean Sea. The range of chromium levels found in the sperm whale samples was much higher than previously reported for wildlife, and was higher in the Pacific and Indian Oceans than in the Atlantic Ocean or Mediterranean Sea. Previous to this study, aluminum and chromium were not considered to be major health concerns. Mercury and selenium were detected in the samples, but mercury levels were not considered to be toxic to the whales (was detoxified under the tiemannite (mercuric selenide) form, and therefore was not

potentially toxic for the whales). Also detected in the samples were lead and cadmium (Ocean Alliance 2010).

Bouquegneau et.al.(1997) studied the toxicological effects of nine heavy metals on the sperm whales stranded on the Belgian coast. Metals studied included, zinc, copper, cadmium, lead nickel iron chromium, selenium and mercury. Metals have been analyzed in the liver, muscle and kidneys of whales. The concentrations of all studied contaminants were low except of mercury and cadmium. The mercury content in tissues was high, but present in the inorganic form. The researchers found the close correlation between mercury and selenium contents of the liver which strongly suggested that the pollutant was detoxified under the tiemannite (mercuric selenide) form, and therefore was not potentially toxic for the whales. Cadmium, however was found in the high concentrations, which is natural and expected due to the diet of this species (cephalopods). On the other hand, cadmium found was not bound to the protein, which is known for its protective effect against toxicity of heavy metals.

Literature suggests that a wide range of natural concentrations of heavy metals can be found in the species of the sperm whale in relation with age, sex, and season (Bouquegneau et.al., 1997). It is believed that heavy metals can be stored and detoxified by marine mammals by binding to the specific proteins, by a compartmentation within lysosomes or by precipitation inspecific granules. Marine mammals are able to bind metals such as zinc, cadmium, copper, and inorganic mercury to low-molecular weight proteins. This is a result of the development of the protective, detoxification mechanisms due to ingestion of large amount of food containing high levels of toxic compounds. As the result of this protective mechanism, the observed concentration of heavy metals in the tissues are nontoxic.

- Blue whale, *Balaenoptera musculus*

According to NOAA (1998), the planktivorous diet of Blue whales makes them less susceptible than piscivorous baleen whales (with fish-based diet) to the accumulation of organochlorine and metal contaminants in their tissue. Researchers point out that there is no reason to suspect that levels of these substances in any baleen whales are presently high enough to cause toxic or other effects (O'Shea and Brownell 1994), although possible long-term or transgenerational impacts remain unstudied.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of heavy metals on any of the ESA-listed whale-species. In addition, the EPA did not find any information which would suggest that concentrations of heavy metals which were adopted by the USVI as water quality criteria would pose a threat to the recovery of any of the ESA-listed whale species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for heavy metals are NLAA the recovery of any of the ESA-listed whale species or their habitats.

Should additional information related to the effects of metals on whale species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

- Hawksbill sea turtle, *Eretmochelys imbricata*

As indicated by NOAA and FWS (1998), Hawksbills depend upon sea grass and/or coral reef habitats for food and habitat. The destruction or degradation of these sensitive ecosystems is a direct threat to the recovery of the ESA-listed species. The general degradation of these habitats can be affected by "chemical poisoning". Organic chemical pollutants, such as petroleum, sewage, pesticides, solvents, industrial discharges, and agricultural runoff are responsible for an sea turtle mortality each year. Environmental contamination also harms biologically important nearshore ecosystems, including seagrass, coral, mangrove, and algae communities. The declining productivity of seagrass and coral communities, in particular, can be hazardous to sea turtles that depend on these systems for nutrition and shelter. However, as NOAA indicated, little is known about threats to foraging populations of hawksbills.

Zinc concentrations in turtles were quite homogenous in the different tissues, compared to mammals. This would suggest a slight different organotropism of these trace elements in turtles compared to seabirds or marine mammals.

The pancreas exhibited high metal concentrations which have never been previously reported in marine vertebrates. This organ has been analyzed in a study carried out by André et al. (1990) about cadmium contamination of tissues of a delphinid species Stenella attenuata. Cadmium average concentration was 5.64 μg/g, much lower than in kidney (48.7 μg/g) or liver (8.7 μg/g).

Food is probably the main source of exposure to heavy metals and other trace elements for marine vertebrates. In marine mammals or seabirds, such concentrations (and also much higher) are often encountered in animals feeding mainly on cephalopods which are known to accumulate Cd in higher levels than fish and are considered to be an important vector of this element to top marine predators (Bustamante et al., 1998). Hatchling appears to be nearly omnivorous but the diets of adults are more specialized and differ among species- leatherback turtles feed on jellyfish and loggerhead turtles are mostly carnivorous (Bowen and Avise, 1995).

A study carried out by Godley et al. (1999) suggests that loggerhead turtles in the eastern Mediterranean Sea do indeed feed upon benthic molluscs and crustacea, at shallow to moderate depths, from both rocky and sedimentary habitats. For both post-pelagic juveniles and adults of these species, study of prey items in other regions of the world have found the diet to be dominated by benthic molluscs, crustaceans and coelenterates (Godley et al., 1999). Although such a study has not been done on the stranded individuals of this study, the postmortem examinations very often revealed plastic fragments mixed with benthic preys in the stomach (Duguy et al., 1998). These food items are not very specific and the high Cd concentrations found in sea turtles could also be the consequence of the metabolism or physiology of these species which would lead to a great accumulation of this element.

In addition, NOAA and FWS (1993) also indicated that pesticides, heavy metals and have been detected in turtles (including eggs), but levels that result in adverse effects have not been

quantified. The specific effects of marine pollution on hawksbills, their eggs, and their prey have yet to be determined.

- Green sea turtle, *Chelonia mydas*

Andreani et al (2008) determined the concentrations of selected trace elements (Zn, Cu, Fe, Mn, Cd and Pb) in tissues of green turtles from Tortuguero National Park on the North Caribbean coast of Costa Rica and of loggerheads from the Mediterranean Sea. Zn, Cu, Fe, Mn and Cd were present at detectable concentrations in all samples and showed clear organotropism, whereas Pb was not always over the detection limit and did not show any particular tissue distribution. The two species presented significant differences: Cu and Cd in liver and kidney of *Chelonia mydas* were significantly higher with respect to the concentrations found in *Caretta caretta*.

Barbieri (2009) analyzed 30 specimens (15 adults and 15 juveniles) of *Chelonia mydas* found in the Cananéia estuary in the state of São Paulo on the southeastern Brazilian coast between January 2005 and September 2006. The concentrations of Cd, Cu, Pb, Mn and Ni in liver and kidney samples of adult and juvenile green turtles were determined by Flame Atomic Absorption Spectrophotometry. The average Cd concentration found in adult livers (0.57µg/g) was significantly higher than that in juveniles (0.279µg/g). Copper concentrations were significantly higher in the liver than in the kidney, and significantly higher in adults (39.9µg/g) than in juveniles (20.7µg/g) Average Mn concentrations in liver and kidney did not differ between adults (4.32 and 4.17µg/g) and juveniles (4.81 and 3.82µg/g), whereas Ni concentrations in adults (0.28 and 0.19µg/g, respectively) were significantly higher than in juveniles (0.13 and 0.089µg/g, respectively). Lead concentrations in liver were significantly higher in adults (0.37µg/g) than in juveniles (0.06µg/g). The concentrations of essential trace elements in *Chelonia mydas* were generally comparable to values reported in other, similar studies. With respect to non-essential metals (Cd, Pb and Ni), *Chelonia mydas* presented lower values than those reported for their northern Atlantic counterparts.

- Loggerhead sea turtle, *Caretta caretta*

Maffucci et al (2005) determined concentrations of Cadmium (Cd), copper (Cu), mercury (Hg), selenium (Se) and zinc (Zn) were in the liver, kidney and muscle of 29 loggerhead turtles, *Caretta caretta*, from the South Tyrrhenian Sea (Western Mediterranean). No significant differences ($p>0.05$) were detected between males and females. Trace element concentrations were not influenced by the size of the specimen except Se in the liver, which was negatively correlated with the curved carapace length ($p<0.001$). Muscles generally displayed the lowest trace element burdens, with the exception of Zn which contained concentrations as high as 176µg/g dwt. Kidneys displayed the highest Cd and Se mean concentrations (57.2±34.6 and 15.5±9.1µg/g dwt, respectively), while liver exhibited the highest Cu and Hg levels (37.3±8.7 and 1.1±1.7µg/gdwt, respectively). Whichever tissue is considered, the toxic elements had elevated coefficients of variation (i.e. from 60% to 177%) compared to those of the essential ones (i.e. from 14% to 65%), which is a consequence of homeostatic processes for Cu, Se and Zn. Globally, the concentrations of Hg remained low in all the considered tissues, possibly the result of low trophic level in sea turtles. In contrast, the diet of loggerhead turtles would result in a significant exposure to Cd. Highly significant correlations between Cd and Cu and Zn in the

liver and kidney suggest that efficient detoxification processes involving MT occur which prevent Cd toxicity in loggerhead turtles.

Franzellitti et al. (2004) collected 35 specimens of *Caretta caretta* were collected dead along the Adriatic Sea coast in Italy. Turtles were classified into four size categories ranging from 24.5 to 74 cm, by measuring the *minimum straight-line carapace length*. Cd, Cu, Fe, Mn, Ni, and Zn levels were assessed in liver, lung, muscle and adipose tissue. Cd, Cu and Fe mainly accumulated in the liver (8.9, 23.7 and 1180 mg/kg dry mass [d.w.], respectively), and Mn in the lung (29.5 mg/kg d.w.). Levels of Ni were higher in adipose (22 mg/kg d.w.) than other tissues, while Zn concentrations were higher in muscle (about 140 mg/kg d.w.). Negative correlations with size were established for Zn in liver and Cu in adipose tissue, while positive correlations were observed for Mn and Ni in adipose tissue. Metal concentrations did not differ between males and females, nor between individuals found stranded and those victims of by-catch. On average, Cd, Cu, Mn and Ni concentrations in our specimens were higher than in loggerhead turtles and other species living in other areas.

Godley et al (1999) determined the concentrations of heavy metals (Hg, Cd and Pb) in internal organs and nest contents of green turtles *Chelonia mydas* and loggerhead turtles *Caretta caretta* from northern Cyprus, eastern Mediterranean Sea. Concentrations of mercury in liver tissue were higher in loggerhead turtles (median 2.41 μg/g dw) than in green turtles (0.55 μg/g dw). Preliminary data suggest cadmium concentrations to be highest in kidney tissue of loggerhead turtles (median 30.50 μg/g dw) but in liver tissue of green turtles (median 5.89 μg/g dw). Concentrations of lead in internal tissues were often below analytical detection limits in both species, but when measurable, tended to be higher in loggerhead turtles. Concentrations of mercury and cadmium in nest contents from both species were low, often below analytical detection limits, while those of lead were relatively high in loggerhead turtle hatchlings (up to 10.56 μg/g dw). When measurable, concentrations of all three metals tended to be higher in loggerhead turtle nest contents than in green turtle nest contents. Results presented here are consistent with inter-specific differences in diet and trophic status. Heavy metal burdens in loggerhead turtles and green turtles from the Mediterranean are similar or lower than corresponding concentrations in turtles from Japan and Hawaii, but some lead concentrations in Mediterranean loggerhead hatchlings are at levels known to cause subclinical toxic effects in other vertebrates.'

Heavy metal concentrations in the tissues of marine turtles were presented by Storelli and Marcotrigiano (2003). The most frequently monitored elements are mercury, cadmium and lead; and the tissues mainly analyzed in nearly all the stranded individuals are muscle, liver and kidney. The highest mercury and cadmium levels were found in liver and kidney respectively; the majority of the lead burden existed in bones and carapace, while arsenic was present mainly in muscle tissue. Mercury occurred quite completely as methylmercury in muscle, whereas in liver the main form was the inorganic one. Arsenic was exclusively present in the metallorganic form either in muscle tissue or in liver. Metals in the eggs were mainly present in the yolk. Significantly higher concentration of mercury, copper, zinc and iron were found in yolk than albumen, while shell contained highest levels of manganese and copper. The load of trace metals in these animals strictly correlated with the species seems to depend on their different food behavior.

Gardner et al (2006) assessed heavy metals in four species of sea turtles from the Baja California Peninsula, Mexico, representing the first report of heavy metal concentrations in tissues of post-yearling sea turtles from the Eastern Pacific. Concentrations of Cd measured in *C. mydas* kidney (653μg/g dw) were the highest ever reported for any sea turtle species. Cd accumulated preferentially in kidney and the ratios of kidney to liver Cd in Baja California turtles were among the highest reported for sea turtles globally. Zn, Ni, and Mn concentrations were also significantly higher in kidney than other tissues, while Cu and Fe were greatest in liver, and all metals were lowest in muscle. With the exception of one value (69.9μg/g in kidney of *C. caretta*), Pb was low in all tissues from Baja California. In comparisons across species, kidney of *C. mydas* had greater Zn and Ni concentrations as compared to other species, although there was no difference in liver metal levels among the species. Positive correlations were detected in the concentrations of Cd, Cu and Ni with the straight carapace length of *C. caretta*.

Saeki et al (2000) determined concentrations of arsenic in the liver, kidney and muscle of three species of sea turtles, e.g., green turtles (*Chelonia mydas*), loggerhead turtles (*Caretta caretta*) and hawksbill turtles (*Eretmochelys imbricata*), were determined using HG-AAS, followed by arsenic speciation analysis using HPLC-ICP-MS. The order of arsenic concentration in tissues was muscle >kidney> liver. Unexpectedly, the arsenic concentrations in the hawksbill turtles feeding mainly on sponges were higher than the two other turtles primarily eating algae and mollusk which accumulate a large amount of arsenic. Especially, the muscles of the hawksbill turtles contained remarkably high arsenic concentrations averaging 153mg/kg dry weight with the range of 23.1–205mg/kg (n=4), even in comparison with the data from other organisms. The arsenic concentrations in the tissues of the green turtles were significantly decreased with standard carapace length as an indicator of growth. In arsenic compounds, arsenobetaine was mostly detected in the tissues of all the turtles. Besides arsenobetaine, a small amount of dimethylarsinic acid was also observed in the hawksbill turtles.

Yasumi et al. determined concentrations of 18 trace elements (V, Cr, Mn, Co, Cu, Zn, Se, Rb, Sr, Zr, Mo, Ag, Cd, Sb, Ba, Hg, Tl, and Pb) in the liver, kidney, and muscle of green turtles (*Chelonia mydas*) and hawksbill turtles (*Eretmochelys imbricata*) from Yaeyama Islands, Okinawa, Japan. Accumulation features of trace elements in the three tissues were similar between green and hawksbill turtles. No gender differences in trace element accumulation in liver and kidney were found for most of the elements. Significant growth-dependent variations were found in concentrations of some elements in tissues of green and hawksbill turtles. Significant negative correlations ($p < 0.05$) were found between standard carapace length (SCL) and the concentrations of Cu, Zn, and Se in the kidney and V in muscle of green turtles and Mn in the liver, Rb and Ag in kidney, and Hg in muscle of hawksbill turtles. Concentrations of Sr, Mo, Ag, Sb, and Tl in the liver, Sb in kidney, and Sb and Ba in muscle of green turtles and Se and Hg in the liver and Co, Se, and Hg in kidney of hawksbill turtles increased with an increase in SCL ($p < 0.05$). Green and hawksbill turtles accumulated extremely high concentrations of Cu in the liver and Cd in kidney, whereas the levels of Hg in liver were low in comparison with those of other higher–trophic-level marine animals. High accumulation of Ag in the liver of green turtles was also observed. To evaluate the trophic transfer of trace elements, concentrations of trace elements were determined in stomach contents of green and hawksbill turtles. A remarkably high trophic transfer coefficient was found for Ag and Cd in green turtles and for Cd and Hg in hawksbill turtles.

Caurant et al (2013) studied heavy metals bioaccumulation in sea turtles using eggs more often than tissues and organs. The purpose of this study was to assess the presence of cadmium, a toxic metal relatively abundant in the Gironde estuary, copper and zinc in different tissues of turtles stranded along French Atlantic coasts. Cadmium, copper and zinc have been analyzed in some tissues and organs of Loggerhead, Kemp's Ridley (only muscle for this species) and Leatherback turtles stranded along the Atlantic coasts of France. The pancreas analyzed only in Leatherback turtles exhibited the highest metal concentrations, which is very surprising for an organ which does not play a role in the detoxification processes. The distribution of these elements in kidney, liver and muscle were quite similar to that found in marine mammals or seabirds. Nevertheless, mean cadmium concentrations in the kidney were as high as 13.3 μg/g wet weight in the Loggerhead turtles and 30.3 μg/g wet weight in the Leatherback turtles. Such high concentrations in the Leatherback turtles have never been recorded before. The main source of cadmium for marine turtles is probably the food. The Leatherback turtles are known to feed mainly on jellyfish in this area. Ten times higher cadmium concentrations have been determined in jellyfish compared to fish. This would imply a greater exposure to cadmium for Leatherback turtles, which probably need to eat great quantities of jellyfish to cover their needs.

Conclusion: During the scientific literature review, the EPA did not come across any research which would report on the specific effects of heavy metals on any of the ESA-listed sea turtle species. In addition, the EPA did not find any information which would suggest that concentrations of heavy metals which were adopted by the USVI as water quality criteria would pose a threat to the recovery of any of the ESA-listed sea turtle species. As a result, the EPA has determined that the water quality criteria adopted by the USVI for heavy metals are NLAA the recovery of any of the ESA-listed sea turtle species or their habitats.

Should additional information related to the effects of metals on sea turtle species become available, the Agency will reevaluate its determination and work with the USVI to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

X. Effects Determinations

This document presented the evaluation of the aquatic life water quality standards adopted by the USVI on September 9, 2015. This Biological Evaluation has been prepared to support the EPA's determination of "not likely to adversely affect" any of the eighteen threatened and endangered marine species located in the U.S. Virgin Islands waters covered by the subject water quality standards actions. The adoption of the 2015 water quality standards by the U.S. Virgin Islands Government and EPA approval of this action are considered to be NLAA based on a holistic consideration of the "best available scientific and commercial data." The EPA views the adoption of numeric water quality criteria as an important step forward for the U.S. Virgin Islands in being able to restore and/or protect the aquatic life within estuaries and coastal environment.

XI. Recommendations to be considered during the next Triennial WQS Review Process

The next triennial WQSR review process is schedule for the 2018. As a result of this biological evaluation, the EPA will work closely with the VI DPNR and NOAA-NMFS on potential revisions of the pH and clarity standards to ensure that all of the ESA-listed species and their habitats are being adequately protected. In addtion, as the additional information related to the effects of any of the water quality parameters discussed in this document on any of the ESA-listed species becomes available, the Agency will reevaluate its determination and work with the VI DPNR to revise the applicable water quality criteria, if needed, to ensure that they are fully protective of this ESA-listed species.

XII. Tables

Table 1. Water quality criteria for saltwater aquatic life based on total ammonia (mg/L) – Criteria Maximum Concentrations (acute values).

Temperature (C deg)						
	10	15	20	25	30	35
pH	**Salinity = 10 g/kg**					
7.0	131	92	62	44	29	21
7.2	83	58	40	27	19	13
7.4	52	35	25	17	12	8.3
7.6	33	23	16	11	7.7	5.6
7.8	21	15	10	7.1	5.0	3.5
8.0	13	9.4	6.4	4.6	3.1	2.3
8.2	8.5	5.8	4.2	2.9	2.1	1.5
8.4	5.4	3.7	2.7	1.9	1.4	1.0
8.6	3.5	2.5	1.8	1.3	0.98	0.75
8.8	2.3	1.7	1.2	0.92	0.71	0.56
9.0	1.5	1.1	0.85	0.67	0.52	0.44
pH	**Salinity = 20 g/kg**					
7.0	137	96	64	44	31	21
7.2	87	60	42	29	20	14
7.4	54	37	27	18	12	8.7
7.6	35	23	17	11	7.9	5.6
7.8	23	15	11	7.5	5.2	3.5
8.0	14	9.8	6.7	4.8	3.3	2.3

8.2	8.9	6.2	4.4	3.1	2.1	1.6
8.4	5.6	4.0	2.9	2.0	1.5	1.1
8.6	3.7	2.7	1.9	1.4	1.0	0.77
8.8	2.5	1.7	1.3	0.94	0.73	0.56
9.0	1.6	1.2	0.87	0.69	0.54	0.44
pH	**Salinity = 30 g/kg**					
7.0	148	102	71	48	33	23
7.2	94	64	44	31	21	15
7.4	58	40	27	19	13	9.4
7.6	37	25	21	12	8.5	6.0
7.8	23	16	11	7.9	5.4	3.7
8.0	15	10	7.3	5.0	3.5	2.5
8.2	9.6	6.7	4.6	3.3	2.3	1.7
8.4	6.0	4.2	2.9	2.1	1.6	1.1
8.6	4.0	2.7	2.0	1.4	1.1	0.81
8.8	2.5	1.8	1.3	1.0	0.75	0.58
9.0	1.7	1.2	0.94	0.71	0.56	0.46

Table 2. Water quality criteria for saltwater aquatic life based on total ammonia (mg/L) – Criteria Continuous Concentrations (chronic values).

Temperature (C deg)						
	10	15	20	25	30	35
pH	**Salinity = 10 g/kg**					
7.0	20	14	9.4	6.6	4.4	3.1
7.2	12	8.7	5.9	4.1	2.8	2.0
7.4	7.8	5.3	3.7	2.6	1.8	1.2
7.6	5.0	3.4	2.4	1.7	1.2	0.84
7.8	3.1	2.2	1.5	1.1	0.75	0.53
8.0	2.0	1.4	0.97	0.69	0.47	0.34
8.2	1.3	0.87	0.62	0.44	0.31	0.23
8.4	0.81	0.56	0.41	0.29	0.21	0.16
8.6	0.53	0.37	0.27	0.2	0.15	0.11
8.8	0.34	0.25	0.18	0.14	0.11	0.08
9.0	0.23	0.17	0.13	0.1	0.08	0.07
pH	**Salinity = 20 g/kg**					
7.0	21	14	9.7	6.6	4.7	3.1
7.2	13	9.0	6.2	4.4	3.0	2.1
7.4	8.1	5.6	4.1	2.7	1.9	1.3
7.6	5.3	3.4	2.5	1.7	1.2	0.84
7.8	3.4	2.3	1.6	1.1	0.78	0.53
8.0	2.1	1.5	1.0	0.72	0.5	0.34
8.2	1.3	0.94	0.66	0.47	0.31	0.24
8.4	0.84	0.59	0.44	0.3	0.22	0.16

8.6	0.56	0.41	0.28	0.2	0.15	0.12
8.8	0.37	0.26	0.19	0.14	0.11	0.08
9.0	0.24	0.18	0.13	0.1	0.08	0.07
pH	**Salinity = 30 g/kg**					
7.0	22	15	11	7.2	5.0	3.4
7.2	14	9.7	6.6	4.7	3.1	2.2
7.4	8.7	5.9	4.1	2.9	2.0	1.4
7.6	5.6	3.7	3.1	1.8	1.3	0.9
7.8	3.4	2.4	1.7	1.2	0.81	0.56
8.0	2.2	1.6	1.1	0.75	0.53	0.37
8.2	1.4	1.0	0.69	0.5	0.34	0.25
8.4	0.9	0.62	0.44	0.31	0.23	0.17
8.6	0.59	0.41	0.3	0.22	0.16	0.12
8.8	0.37	0.27	0.2	0.15	0.11	0.09
9.0	0.26	0.19	0.14	0.11	0.08	0.07

XIII. Abbreviations

AChE	acetylcholinesterase
ACR	Acute-chronic ratio
AWQC	Ambient water quality criteria
BCF	Bioconcentration factor
BE	Biological Evaluation
BCG	biological condition gradient
CCC	Criteria chronic concentration
CCRP	Caribbean Coral Reef Partnership
CGP	Construction general permit
CMC	Criteria Maximum Concentration
CWA	Clean Water Act
DPS	Distinct Population Segments
EC50	Effective Concentration, the concentration of a chemical that is estimated to produce a specific effect in 50% of the test organisms
EPA	Environmental Protection Agency
EMAP	Environmental Monitoring and Assessment Program
ESA	Endangered Species Act
EQB	Environmental Quality Board
FACR	Final Acute Chronic Ratio
FAV	The final acute value
FCV	The final chronic value
FDA	U.S. Food and Drug Administration
FR	Federal Register

FRV	Final Residue Value
FWS	Fish and Wildlife Service
GMAVs	genus mean acute values
GMCVs	genus mean chronic values
IC50	Inhibitory Concentration, the concentration of a chemical that is estimated to inhibit some biological process (i.e. growth, etc.) by 50% compared to a control organism
IPCC	International Panel on Climate Change
IWC	International Whaling Commission
IUCN	International Union for the Conservation of Nature
LC50	Lethal Concentration, the concentration of a chemical that is estimated to kill 50% of the test organisms
LOEC	Lowest-Observed-Effect-Concentration, is the lowest test concentration at which observed effects were statistically different from the control
MATC	Maximum Acceptable Toxicant Concentration, is the calculated geometric mean of the NOEC and LOEC.
MoA	Memorandum of Agreement
NLAA	Not likely to adversely affect
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOEC	No-Observed-Effect-Concentration, is the highest test concentration at which none of the observed effects were statistically different from the control
NTU	nephelometric turbidity unit
ORD	Office of Research and Development
PAH	polycyclic aromatic hydrocarbons
PCB	polycholorobiphenyls
PRDNER	Puerto Rico Department of Natural and Environmental Resources
PR EQB	Puerto Rico Environmental Quality Board

SMAVs	species mean acute values
SSD	species sensitivity distribution
TEDs	turtle excluder devices
TPDES	Territorial Pollution Discharge Elimination System
USCRI	United States Coral Reef Initiative
USCRTF	United States Coral Reef Task Force
USFWS	United States Fish and Wildlife Service
UVI	University of the Virgin Islands
WWF	World Wide Fund
WQS	Water quality standards
WQSR	Water Quality Sandards Regulations
VIDPNR	Virgin Islands Department of Planning and Natural Resources

XIV. References

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